Rangeland degradation around water-points under different management systems

by

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Declaration

I, the undersigned, hereby declare that the work contained in this thesis is my own original work and that I have not previously in its entirety or part submitted it at any university for a degree.

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Date: Tuesday, 9 March 2004

Abstract

Over 70% of South Africa is too arid for crop farming and is used for commercial livestock ranching, communal livestock ranching, or game ranching. These management systems differ from each other in a number of aspects, e.g. herbivore species, stocking rate, grazing regime, and management structure. The main question addressed in this study is: to what extent do the three management systems affect rangeland condition? Following the equilibrium paradigm of vegetation dynamics, communal livestock ranches are expected to have a greater detrimental effect on rangeland condition than other management systems because stocking densities and, consequently, herbivore impacts, are usually far higher than under the other two management systems. However, recent non-equilibrium theories argue that vegetation dynamics in arid systems are mainly driven by rare and stochastic rainfall events. Biotic factors, such as grazing and herbivore diversity, are predicted to have little effect on rangeland condition. This leads to the prediction that herbivore impacts on arid ecosystems are not density-dependent and, consequently, the high stocking densities commonly recorded on communal livestock ranches will not cause more rangeland degradation than other management systems.

Arid and semi-arid rangelands are characterised by high inherent spatial and temporal variation in vegetation and soil parameters. Hence, differentiating between the effects of grazing management systems and natural variability caused by abiotic factors, such as rainfall, is difficult. This problem can be circumvented by examining gradients of grazing intensity radiating from water-points (= *piospheres*). I examined changes in vegetation and soil parameters along 500 m-long grazing gradients on ranches in the semi-arid Northern Cape province, South Africa, with the three above-mentioned management systems. Analysis of the vegetation on these ranches demonstrated the existence of grazing gradients around the water-points. Two characteristic zones described in other studies were identified, namely the *sacrifice zone* (almost complete denudation of the vegetation close to the water-point) and the *dense shrub zone* (distal to the sacrifice zone). Communal livestock ranching had a greater negative effect on rangeland vegetation than the two other ranching systems, particularly with regard to annual grass species diversity, percentage of bare soil, and shrub density. Within 100 m of the water-points, soil quality was negatively affected by commercial cattle ranching. Contrastingly, there were no significant effects of the other two management systems on soil quality in the piosphere. The feeding of supplemental forage and nutrients may explain the greater impact of

commercial cattle farming on soil quality. The results of this study are consistent with the predictions of the equilibrium theory of vegetation dynamics because grazing impacts were density-dependent in these semi-arid rangelands. However, the results presented here also indicate that forage supplementation alters the spatial and temporal distribution of livestock in a manner that may have greater negative impacts on soil quality than density of livestock *per se*.

Opsomming

Meer as 70% van Suid-Afrika is te droog vir gewasverbouing en word gebruik vir kommersiële veeboerdery, gemeenskaplike veeboerdery of wildboerdery. Die bestuur van diè verskillende boerderytipes verskil wesentlik van mekaar in verskeie opsigte, ondermeer herbivoorspesies, vee-aanskaffingstempo, weidingstipe en bestuurstruktuur. Die belangrikste vraag wat aangespreek word in die studie is die volgende: Tot watter mate affekteer die drie verskillende boerderybestuurstipes die toestand van weivelde? Gesien teen die agtergrond van die ekwilibriumparadigma van plante-dinamika, word verwag dat gemeenskaplike veeboerdery 'n groter impak op die toestand van weiveld sal hê as ander boerderytipes, omdat veedigtheid en gevolglik herbivoor-impak gewoonlik veel hoër is as met die ander twee boerderytipes. Onlangse nie-ekwilibria-teorië voer egter aan dat plante-dinamika in ariede gebiede gedryf word deur seldsame en buitengewone reënvalgebeure. Biotiese faktore, soos weiding en herbivoor diversiteit sal na verwagting min effek hê op die toestand van weiveld. Hieruit volg die voorspelling dat herbivoorimpak op ariede ekosisteme nie afhanklik is van digtheid nie, en dat hoë veedigtheid algemeen aangeteken vir gemeenskaplike boerderye, nie rede sal wees vir groter verliese in weiveld nie.

Ariede en semi-ariede weivelde word gekenmerk deur 'n hoë ruimtelike en tydelike veranderinge in plant- en grondparameters. Gevolglik is dit moeilik om te onderskei tussen die effekte van weidingsbestuur en natuurlike veranderings wat veroorsaak word deur nie-biotiese faktore, soos reënval. Diè probleem kan oorbrug word deur gradiënte van weidingsintensiteit, wat vanaf waterpunte uitstraal, te ondersoek. Vervolgens het ek binne al drie bg. boerderybestuursisteme veranderinge in plant- en grondparameters langs 500m-lange weidingsgradiënte ondersoek op plase in die semi-ariede Noord-Kaap provinsie van Suid-Afrika. Plantegroei-analise het getoon dat 'n weidingsgradiënt rondom waterpunte voorkom. Twee kenmerkende sones is in ander studies geïdentifiseer, nl., 'n opofferingsone (waar amper alle plantegroei naby waterpunte uitgetrap is) en die digte, struik-sone (distaal to die opofferingsone). Gemeenskapsveeboerdery het 'n groter negatiewe effek op plaasgewasse as die ander twee boerderytipes, veral waar dit betrekking het op grasspesie-diversiteit, persentasie grond sonder plantbedekking en struikdigtheid. Binne 100m vanaf waterpunte, is grondkwaliteit merkbaar negatief geaffekteer deur kommersiële veeboerdery. Daarteenoor, was daar geen merkbare effek op die grondkwaliteit naby waterpunte van die ander twee boerderytipes nie. Aanvullingsvoer kan moontlik die rede wees vir die waargenome, groter impak van kommersiële

veeboerdery op grondkwaliteit. Die resultate van die studie strook met voorspellings van die ekwilibria-teorie van plante-dinamika, daarin dat weidingsimpak digtheidsafhanklik is in hierdie semi-ariede plaasgebiede. Die resultate toon ook egter, dat voedingsaanvullings die ruimtelike en tydelike verspreiding van vee in so 'n wyse verander, dat dit moontlik 'n veel groter negatiewe uitwerking op grondkwaliteit het as veedighteid *per se*.

Contents

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Foreword

This study aims at detecting and comparing rangeland degradation caused by three different management systems (i.e. commercial cattle ranching, communal ranching and game ranching) in a semi-arid rangeland near Kimberley, South Africa. Detecting rangeland degradation in semi-arid rangelands is a contentious issue. Much of the debate around the issue of rangeland degradation focuses on the dynamics of vegetation change. One of the objectives is therefore to test whether grazing gradients radiating from water-points are suitable to determine the dominant drivers of vegetation change (vegetation dynamics) in a particular rangeland. Vegetation dynamics must be well understood in order to be able to predict vegetation change. Understanding interactions between rangeland management actions and vegetation change will enable rangeland managers to make better decisions related to rangeland management. We compared the effects of grazing gradient and management type on several vegetation and soil parameters. This will enable us not only to determine the dominant drivers of vegetation change but also the effect of different management systems on rangeland condition.

This thesis consists of four chapters. Chapter 1 is an introduction into the topics related to the study. These include an introduction to the ecology of South African rangelands, vegetation dynamics, carrying capacity concept, rangeland vegetation and soil degradation in relation to herbivory, and the objectives of this study. Chapters 2 and 3 follow the format of the African Journal of Range and Forage Science and the Journal of Arid Environments, respectively. Chapter 2 discusses the part of the study that focuses on rangeland vegetation. Vegetation data are analysed and discussed in order to detect rangeland degradation caused by the three management systems. Chapter 3 analyses and discusses the soil data collected along the above-mentioned transects. The structure of the two articles differ slightly in order to confirm to the style of the journals involved. Chapter 4 contains a brief summary of the main conclusions of the study. Here, we also give management recommendations based on our results and conclusions. Furthermore, we briefly discuss our experience with the research methodology.

Introduction

Contents

1. South African rangelands

1.1. Economic importance of South African rangelands

85% of South Africa's total land surface is not suitable for arable ranching. Of the 85%, Tainton (1999) estimates that 60% is suitable for livestock or game ranching. South Africa's department of agriculture estimates that 59% of South Africa's total land area is only suited for livestock and game ranching (Anonymous, 2002). Most of the rangelands in South Africa are used for cattle, goats, sheep and game (Scholes & Walker, 1993). Rangelands are therefore important to South Africa ecologically as well as economically.

As markets change, the agriculture sector has to adapt and supply products and services that the markets demand. The number of game ranches in South Africa has been increasing rapidly; since 1993 the sector has been growing with 6.75% per year (Kieser, 2001; Tomlinson *et al.*, 2002). The dramatic increase in the number of game ranches in South Africa has largely been caused by the financial difficulties experienced by commercial livestock ranches. Others have seen opportunities in game ranching because of higher returns from game (especially hunting and eco-tourism), perceived lower inputs (especially less labour), and increased numbers of visitors to South Africa since the fall of apartheid. Nonetheless, commercial livestock ranching remains an important economic activity in South African rangelands. South Africa's red meat industry contributes approximately 12 % to the gross value of agricultural products produced in South Africa (Anonymous, 2002). In short, rangelands are economically important to South Africa, partly for their production of agricultural products and partly as tourist destination.

1.2. Rangeland management

South Africa's rangelands are managed in a number of ways:

1. Commercial livestock ranching.
2. Communal livestock ranching.
3. Commercial game ranching.

These three management systems differ in objectives, ownership, management and livestock or game diversity (Table 1). The differences in management characteristics bring along different problems and opportunities.

1.2.1. Commercial livestock ranching

Commercial livestock ranching is a well-developed industry in South Africa. Commercial livestock ranch managers are mostly well educated and much research is done on technical and economical aspects of the industry. Commercial livestock ranches aim to produce a single product that is sold on the market (Tomlinson *et al.*, 2002). Aiming at producing one product for markets demands a high and stable quality product. This is only guaranteed if livestock can continuously feed on nutritious fodder. If the rangeland cannot produce the needed nutrients ranches must supplement nutrients and energy to animals to keep them fit (Meissner, 1999). Supplementing livestock with nutrients and energy is, however, costly and cuts profits. Commercial livestock ranches therefore need to be able to rely on good quality rangeland that provides livestock with most nutrients and energy they need. In order to do this, most commercial livestock ranches in South Africa use a rotational grazing system (Tainton *et al.*, 1999) and generally apply conservative (lower than advised) stocking rates (see Ward *et al.*, 2000). Rotational systems move livestock to another camp as soon as the vegetation does not provide enough nutrients and energy.

Table 1: *Three different management systems and their characteristics.*

	Commercial livestock	Communal livestock	Game
Objectives	Oriented on agricultural markets	Subsistence oriented	Oriented on tourism industry
Ownership	Private	Communal	Private
Management structure	Single manager	Multiple managers	Single manager
Animal diversity	Single livestock species	Many different livestock species	Many different game species

Even though many commercial livestock ranches use low or advised stocking rates and rotational grazing systems, the management system has number of potentially negative effects on rangeland ecosystems. Due to the provision of supplementary feed, stocking rates can be higher during serious droughts than a rangeland can cope with. Rangelands often consist of different vegetation types, e.g. shrublands, savannas, and grasslands. However, cattle are grazers and are not able to make optimal use of shrublands or trees in savannas, which often makes commercial livestock ranches inefficient (Tainton & Hardy, 1999).

1.2.2. Communal livestock ranching

Due to pre-1994 apartheid laws, development of the majority of South Africans was hindered. Today, many of these previously-disadvantaged South Africans are still poverty-stricken and have little opportunity for investment or good education, wages are low and unemployment is high. Therefore, many rural South Africans have some livestock on communal lands to compensate for a lack of income or low wages.

Communal livestock ranches are characterised by many different people owning a number of livestock that graze or browse a communal rangeland. This makes communal livestock ranches very different from either commercial livestock ranches or game ranches. The problem of having multiple managers on a rangeland was considered to contribute to the 'tragedy of the commons' (Hardin, 1968). In the 'tragedy of the commons' it is reasoned that it is more profitable for an individual to overstock the commons (communal lands in England) rather than being concerned about the sustainability of his actions and the condition of the commons. Due to this phenomenon, a growing number of livestock will graze or browse the rangeland and eventually exceed its theoretical carrying capacity, which may lead to rangeland degradation (Hardin, 1968). However, communal livestock owners may be able to decide how large their herd is but cannot link this to a particular grazing strategy because: (1) they are bound to written and unwritten rules that govern the use of the communal rangeland, and (2) because other livestock owners make use of the same piece of land (Tapson, 1993). Unlike commercial livestock ranches, communal livestock ranches rarely use rotational systems but use continuous grazing systems instead (Everson & Hatch, 1999). In these systems, livestock remain on the same rangeland throughout the year. Due to their financial situation, many communal livestock owners do not have the possibility to buy supplementary feeds, which may result in malnutrition or death of animals during dry years. Animals are, however not intended to be

sold on markets. Most products are for their own consumption and only surpluses are sold on markets (Tomlinson *et al.*, 2002). Animals are used for a wide array of purposes, e.g. milk production, social exchange (lobola), slaughter, and draft (Tapson & Rose, 1984).

1.2.3. Game ranches

Recently, game ranches for venison production, trophy hunting and ecotourism have become another major industry on South African rangelands (Kieser, 2001; Tomlinson *et al.*, 2002). Game ranches aim to provide the best services to hunters or tourists in the form of a great diversity of animals. These ranches have continuous grazing with a multitude of different species that include browsers and grazers (Grossman *et al.*, 1999). Game ranching can lead to the creation of grazing gradients around water-points due to trampling. These grazing gradients are however not as serious in areas with mainly 'water-independent' game, i.e. game that can go without water for long periods (Grossman *et al.*, 1999).

2. Rangeland vegetation dynamics

How do herbivores affect rangeland ecosystems? This is a question too complex to answer in this study. It is, however, an important question to discuss because every rangeland manager needs to know what potential negative effects livestock or game may have on the productivity of a rangeland. Knowledge of rangeland dynamics is the key to explain how herbivores affect rangelands. Rangeland dynamics can be described as processes and patterns of change in response to certain events. These events can be any kind of event that affect vegetation abundance and composition, such as rodents making burrows, grazing of livestock and wildlife, fire, or rain. This study mainly focuses on vegetation changes induced directly or indirectly by livestock and wildlife. Examples are grazing, trampling, defecation, urination, etc. It is important to understand rangeland dynamics and be able to predict the effects of various events on rangeland ecosystems in order to prevent undesirable changes through certain management actions or strategies (Dregne, 1983; Behnke & Scoones, 1992; Holechek *et al.*, 1995; Wiegand & Milton, 1996; Tainton & Hardy, 1999; Archer & Stokes, 2000).

There are two main theories of rangeland ecosystem dynamics. These theories hypothesise that:

1. Rangeland ecosystems are potentially stable (equilibrium) systems and develop towards a stable vegetation, also called a domain of attraction or climax vegetation (Clements, 1916). Vegetation and herbivore population dynamics influence each other.
2. Rangeland ecosystems are unstable (non-equilibrium) systems. Vegetation and herbivore populations are strongly influenced by external forces such as precipitation (Ellis & Swift, 1988; Behnke & Scoones, 1992).

Rangeland dynamics might best be described by one of these two theories, although it is more likely that ecosystems exist along a continuum from equilibrium conditions to non-equilibrium conditions (Wiens, 1984) (Figure 1). The position of a rangeland on the continuum is determined by the extent to which a system is structured by biotic or abiotic controls (Wiens, 1984). Equilibrium systems are thought to be modified by biotic factors and in non-equilibrium systems biotic modifiers have minor influence on vegetation dynamics.

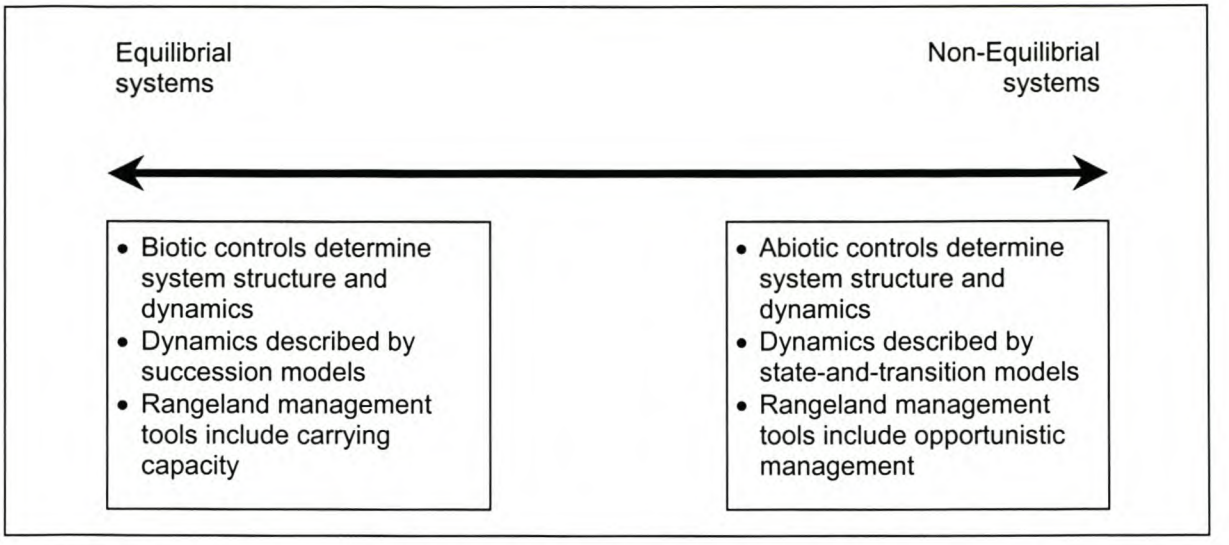

Figure 1: *Ecosystems exist on a continuum between equilibrium and non-equilibrium conditions* (Wiens, 1984).

Models discussed here aim to predict events on small scales, for management units such as grazing camps or ranches. Models can therefore assist in management decisions. There are two important models that are frequently used to describe rangeland ecosystems:

1. successional models;
2. state-and-transition models.

The models are quite different from each other in the way they perceive how a system changes. Change is perceived to occur either predictability for successional models or stochastically for state-and-transition models (Milton *et al.*, 1994). In the following sections, the theories and models discussed above will be discussed in more detail.

2.1. Equilibrium ecosystems

Equilibrium theories hypothesise that ecosystems undergo regular grazing, fire, and drought disturbances (Walker & Noy-Meir, 1982) and that biotic events, such as grazing, have significant effects on vegetation development and composition (Noy-Meir, 1982; Holechek *et al.*, 1995; Illius & O'Connor, 1999; Tainton *et al.*, 1999). Under a stable climate and in the absence of disturbances ecosystems develop towards an equilibrium vegetation state often called climax vegetation (Clements, 1916). Certain events may change the vegetation but when the disturbance is taken away the vegetation redevelops towards the equilibrium vegetation state. Another important characteristic of equilibrium systems is that herbivore-plant dynamics are coupled and follow density dependant

models. Successional models aim to describe these ecosystems; details of these models are described in following sections.

2.1.1. Successional models

The succession model (Clements, 1916) was one of the first models that was applied to describe rangeland ecosystem dynamics, and has since then dominated the way rangeland managers view rangeland ecosystem dynamics (Ellis & Swift, 1988; Joyce, 1993; Holechek *et al.*, 1995, Tainton & Hardy, 1999). The original succession model of Clements is based on the notion that vegetation in a particular area with similar climatic conditions develops from one stage to another to reach a final vegetation type, the climax (Clements, 1916; Clements, 1936). Odum (1971:251) defines succession "... following three parameters: (1) succession is an orderly process of community development that involves changes in species structure and community processes with time; it is reasonably directional and, therefore predictable. (2) It results from modification of the physical environment by the community; that is, succession is community-controlled even though the physical environment determines the pattern, the rate of change, and often sets the limits as to how far development can go. (3) It culminates in a stabilized ecosystem in which maximum biomass (or high information content) and symbiotic function between organisms are maintained per unit of available energy flow." Based on these parameters rangeland scientist and managers developed methods to keep rangeland in an optimal grazing state, or ideal grazing vegetation. For semi-arid rangelands, the ideal grazing vegetation is often considered to be the climax vegetation or near climax vegetation (Tainton & Hardy, 1999); management therefore aims to stimulate succession and minimise events that cause retrogression (Tainton & Hardy, 1999). Succession is the development of a vegetation towards the climax (also called domain of attraction). The main events that stimulate succession in rangelands are resting, reducing stocking densities (biotic events, events that can be controlled by the rangeland manager), and rainfall (abiotic event) (Tainton *et al.*, 1999). Retrogression is the inverse of succession, retrogression is caused by events that cause vegetation to return to an earlier successional stage, this can be caused by events such as continuous heavy grazing (biotic event), fire and drought (abiotic events) (Holechek *et al.*, 1995). Figure 2 shows the ideal grazing vegetation (circle), imagine the vegetation is continuously heavily grazed by cattle, the successional model predicts that this event will cause regression to an earlier vegetation state less suitable for grazing.

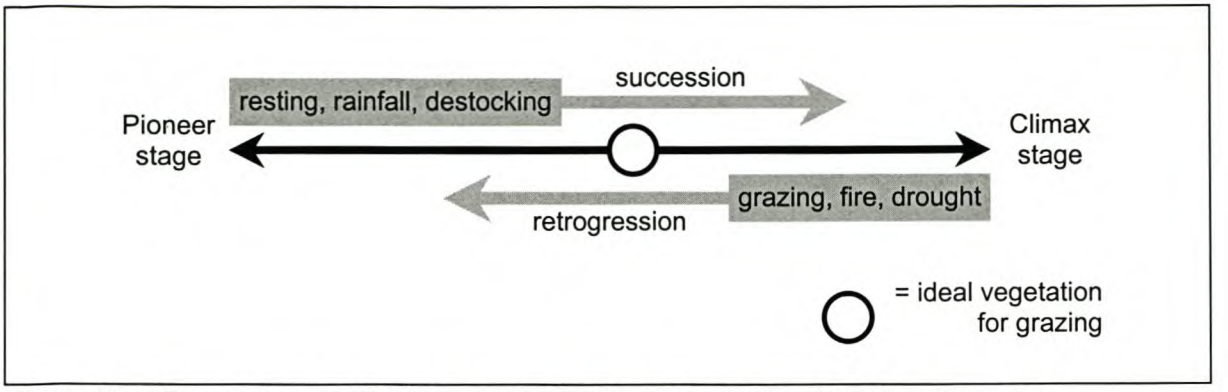

Figure 2: *Succession model where grazing causes regression and resting succession* (Westoby *et al.*, 1989a).

When the area is not disturbed succession forces the vegetation to develop through the same path towards the climax vegetation, palatable species regain there position in the system as they are not grazed and the vegetation reaches the stage it was in before.

2.1.2. Criticism of equilibrium theory

The equilibrium theory regards "…succession as relatively well ordered and vegetation changes reasonably predictable" (Tainton & Hardy, 1999:11). Criticism has been raised towards the idea that succession is relatively ordered, because vegetation change due to an event such as grazing of cattle does not always lead to a vegetation that occurred in an earlier succession phase. For example, Ellison (1960) suggests seven positive effects that grazing may have on the vegetation:

1. stimulation of herbage production;
2. increase drought endurance of plants;
3. increased spring production due to removal of mulch;
4. seed dispersal by animals;
5. trampling helps plant seeds;
6. livestock trails follow contours, therefore increasing water infiltration;
7. excrement and decay of their bodies fertilise the soil.

These positive effects may cause different reactions to herbivore density other than degradation.

Another concern is that the early successional models considered that ecosystems always develop towards a single climax formation (also called domain of attraction), the so-called globally stable systems (Walker & Noy-Meir, 1982). These type of systems are rare and it is more likely that ecosystems have more than one domain of attraction (Walker & Noy-Meir, 1982). Walker & Noy-Meir

(1982) propose that many ecosystems have "multiple equilibria" or "multiple stable states". Multiple stable states have been identified in many ecosystems (see Rietkerk & Van de Koppel, 1997; Van de Koppel *et al.*, 1997; Gunderson, 2000), including arid and semi-arid rangelands. In savannas, two characteristic vegetation states occur; one where grass dominates and in the other trees. These two states are thought, in the Serengeti, to be caused by the occurrence of intense fires and elephants (Dublin *et al.*, 1990). The transition of woodland to grassland occurs after an intense fire, where after elephants keep the newly established grasslands shrub and tree free, this state may then remain stable for a long period. When the elephant population decreases shrubs and trees are able to establish themselves and turn the vegetation into a woodland. As in the previous example, bush encroachment may also be a form of multiple stable state in arid and semi-arid rangelands (Scholes & Walker, 1993).

In addition to the above-mentioned points, the climax theory has also been widely criticised on the idea that climate alone controls a climax (Joyce, 1993). Nevertheless, this theory is still applied by many rangeland managers and agricultural researchers (Ellis & Swift, 1988; Westoby *et al.*, 1989; Hoffman, 2000).

2.2. Non-equilibrium ecosystems

Some rangeland ecologists argue that certain ecosystems, such as arid and semi-arid rangelands are non-equilibrial systems (Ellis & Swift, 1988; Westoby *et al.*, 1989a, 1989b; Behnke & Scoones, 1992, 1993). Non-equilibrial ecosystems are systems that do not reach a stable vegetation state. These systems are "... strongly influenced by external forces rather than, or in addition to, internal biotic factors" (Ellis & Swift, 1988:453). Rainfall in arid and semi-arid rangelands is often erratic (Noy-Meir, 1973). This causes unpredictable growing conditions that are needed for equilibrium ecosystems (Ellis & Swift, 1988). Any influence biotic factors may have, such as controlling plant biomass through grazing, are much smaller than the influence of abiotic forces as rainfall. Traditional rangeland management is not suited for ecosystems that have non-equilibrium dynamics; for example, the use of carrying capacities has little use in these ecosystems. Carrying capacities are calculated to determine the optimal number of herbivores for a particular area under a particular management strategy (see carrying capacity section). Ward *et al.* (1998) studied two ranches in an arid area of Namibia, one commercial cattle ranch with conservative stocking levels and one communal cattle ranch with stocking rates that exceeded carrying capacity. Although there was a large difference in

both management regime and stocking level, no difference in rangeland degradation could be found. Here the use of carrying capacity is not supported by the argument that high stocking levels have a detrimental effect on rangeland ecosystem.

Non-equilibrium theories have received increasing attention and have been discussed and tested by several rangeland scientists (e.g. Tapson, 1993; Milton & Hoffman, 1994; Ward *et al.*, 1998; Fernandez-Gimenez & Allen-Diaz, 1999; Hoffman, 2000). A model that incorporates some of the ideas of the non-equilibrium theory to analyse and predict ecosystem response to certain events is the state-and-transition model (Westoby *et al.*, 1989a; Milton & Hoffman, 1994 – see below).

2.2.1. State-and-transition model

Unlike the succession model, the state-and-transitional model predicts that a vegetation state exists as long as conditions allow it to do so. Certain events can cause a transition towards a new vegetation state which might persist for a long time until another event or combination of events occurs (Westoby *et al.,* 1989a). Westoby *et al.* (1989a) describe five mechanisms that may indicate that an ecosystem has the characteristics of a state-and-transition system:

1. some plant populations may require a rare event for establishment to occur;
2. plant abundance may vary discontinuously and irreversibly in response to changes in stocking rate;
3. alternative stable states may result when the outcome of competition depends on the initial abundance of the competitors;
4. some vegetation components promote fire and are also themselves promoted by fire in the seedling stage;
5. a vegetation change that triggers a persisting change in soil conditions may not be reversible on a timescale relevant to management.

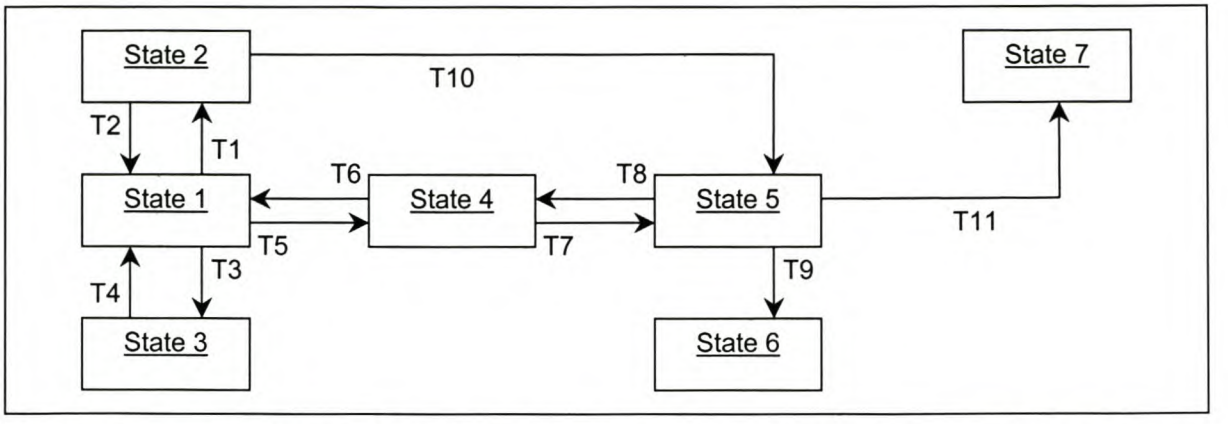

Figure 3: *State-and-transition model. States may be irreversible and triggered by transitions. States are shown as boxes and transitions are shown as arrows* (Milton & Hoffman, 1994).

They argue that each of these mechanisms can produce alternative vegetation states that are stable and irreversibly.

Successional models develop towards one stable vegetation state, while the state-and-transitional model can develop into a myriad of vegetation states depending on the events that precede the change. The model can be visualised as in Figure 3. A complete model should give a description of the vegetation composition of the states and a description of the events that trigger the transition with a description of the intermediate transition vegetation. Laycock (1991) and Milton & Hoffman (1994) present state-and-transition models for North American sagebrush grass ecosystem and southern Karoo shrublands, respectively.

2.2.2. Criticism of non-equilibrium theory

Illius & O'Connor (1999) questioned the non-equilibrium concept on several points:

1. Are there cases where the lack of equilibrium results from a fundamentally different dynamic regime ("non-equilibrium dynamics") from where density-dependent regulatory mechanisms cause the system to tend towards equilibrium?
2. Are there special properties of fluctuating populations that give rise to important community phenomena?
3. Do communities subject to environmental variability merely form a continuum according to strength and frequency of perturbation, but without any characteristically different form of consumer-resource relations?

Ellis & Swift (1988) argue that the variability in the availability of forage in semi-arid rangelands is so high, due to low and stochastic rainfall, that it can never lead to density-dependent regulation of herbivores. Illius & O'Connor (1999) contest this and argue that herbivores in semi-arid rangelands are regulated by density-dependent processes during the dry season when forage is minimal. In this period, the herbivore population is regulated by the availability of forage from key resources. Spatial heterogeneity in vegetation and heterogeneity in nutritional quality of the vegetation creates these key resources. An area which retains water for a longer period, such as drainage lines, may be a preferred resource for herbivore populations and, therefore, a key resource. During dry periods, availability of forage in these key resource areas limits herbivore population. This creates a density-dependent relationship between the key forage resources and the herbivores using these key forage resources.

The question remains how vegetation in the key resources is affected by herbivory. The non-equilibrium theory hypothesises that vegetation dynamics in non-equilibrium systems are driven by stochastic abiotic events, such as rainfall, drought, and fire (Westoby *et al.*, 1989). Again Illius & O'Connor (1999) question these ideas, they present several studies that showed that grazing did have an impact on the vegetation of semi-arid rangelands. Illius & O'Connor (1999) continue to explain that there is a difference in the effect of abiotic events and biotic events. Abiotic events can have negative effects on productivity of a rangeland, e.g. drought decreases plant productivity, or other abiotic events, such as rain and fire, may actually increase productivity. For example, fire is often used to increase nutritional value of the vegetation (Trollope, 1999). The net change due to abiotic events, is therefore minimal over the long term (Illius & O'Connor, 1999). However, biotic events predominantly have a consistent effect on a given plant species (O'Connor & Roux, 1995). This results in a cumulative and substantial change over the long-term (Illius & O'Connor, 1999).

3. Carrying capacity

The focal point of rangeland management is the control of livestock (Holechek *et al.*, 1995; Tainton *et al.*, 1999). Manipulation of the number of livestock in a herd, the time that a herd grazes a camp, and when a herd grazes a certain camp is all part of rangeland management. All these philosophies stem from the idea that herbivores affect rangeland vegetation. Limiting herbivores to a certain herd size and grazing period per camp aims to sustain a healthy herd and healthy vegetation (Stoddart, 1960). Models of density-dependence are models that assume that there is a negative relationship between herbivore density and plant-available biomass. These models are used to predict the number of herbivores per hectare without causing irreversible degradation to the vegetation, moreover they are used to calculate herbivore-density for optimal animal-production (Stoddart, 1960). This section will discuss these models.

3.1. Density-dependence models

Density-dependent models aim to describe the relationship between the different biotic parameters, and predict a coupling of vegetation biomass and herbivore density. Figure 4 is a simplified graph by

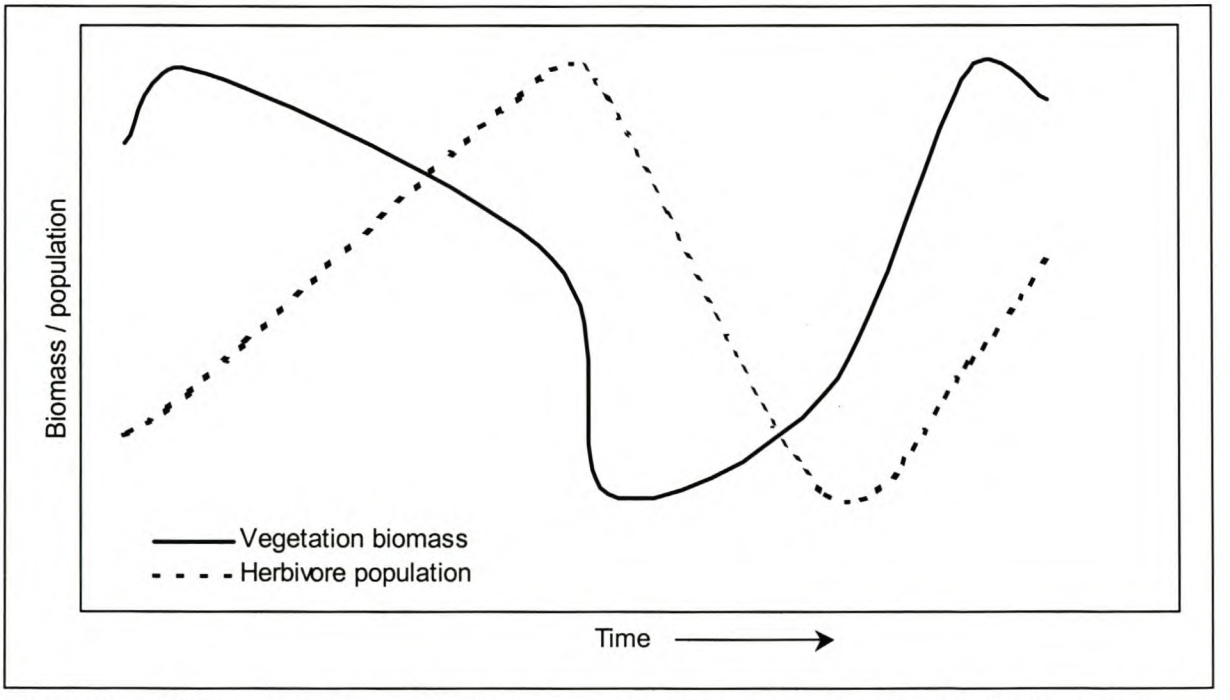

Figure 4: *Time course of vegetation biomass and herbivore population in a density-dependant model* (Noy-Meir, 1982:569).

Noy-Meir (1982) of such a density-dependent model. The graph shows how the increase in vegetation biomass declines as the herbivore population grows. At a certain point, the vegetation biomass collapses and is dramatically reduced. Herbivores have reserves to survive average drought periods in which forage is limited. Therefore, the herbivore population does not react immediately to the vegetation crash by a decrease in herbivore population, but it does when reserves start to become depleted. When the herbivore population collapses, the vegetation starts to regain biomass and the herbivore population follows with a positive growth trend. These models are called density-dependent models as vegetation density and herbivore density are dependent on each other. Noy-Meir (1982) adds that this model assumes that growth parameters (rain, temperature, radiation) are constant; vegetation is considered to be equally palatable; herbivores are homogenous; and that there are no feedback mechanisms from vegetation or herbivores to the environment. In reality, of course, these variables may vary considerably. More complex models that consider vegetation-herbivore density dependence have been discussed by Noy-Meir (1982).

3.2. Carrying capacity

Carrying capacity is a much-debated concept in rangeland management and science literature (e.g. Stoddart, 1960; Behnke & Scoones, 1992; Abel, 1993; Bartels *et al.*, 1993; De Leeuw & Tothill, 1993). There is no overall agreement on the definition of carrying capacity. Bartels *et al.* (1993) reviewed 12 definitions of the carrying capacity concept to determine the necessary elements of a good general definition. They concluded that such a definition needs to encompass elements of animal production goals, acceptable resource condition, and time. Of the 12 definitions, the definition that encompasses all these elements was proposed by Giles (1978:195): *carrying capacity* is "... the user-specified quality biomass of a particular species, under the influence of behavioural constraints, for which a particular area, having user-specified objectives, will supply all energetic and physiological requirements over a long (but specified) period."

Estimation of carrying capacity can be based on many factors, but three factors are important in the calculation: (1) fodder availability and its quality throughout a year; (2) secondary production goals, i.e. the benefit from the available fodder in relation to the character or level of secondary production considered; (3) sustainability of the system, i.e. the fraction which can be used and still maintain future availability of fodder and its quality (De Ridder & Breman, 1993:113).

Carrying capacity was developed to "... determine the numbers of animals which will give the maximum meat and wool yields and yet not endanger soil and water stability nor unduly interfere with other land-uses" (Stoddart, 1960:251). When we examine this objective and other carrying capacity definitions, discussed above, we see that one of the main objectives is to prevent rangeland degradation and production loss. The main idea behind the carrying capacity concept is that there is an optimal and maximum number of animals that can graze or browse a rangeland without causing degradation. Many different carrying capacities can be calculated, of which the economic carrying capacity (the *maximum sustained yield sensu* Behnke & Scoones, 1992), and the ecological carrying capacity (the situation where herbivore population is in equilibrium with the vegetation (Behnke & Scoones, 1992)) are the most important. Exceeding the economic carrying capacity leads to a loss in production due to lower herbivore population growth, whereas exceeding the ecological carrying capacity leads to long-term degradation of the rangeland due to sustained high herbivore pressure and all the negative affects linked to that. Under natural conditions, livestock populations will not be able to exceed the ecological carrying capacity due to limited food resources. Ranches, however, often give cattle roughage and nutrient supplements, which enables them to sustain high cattle populations.

3.3. Applicability of the carrying capacity concept

Not only is the definition of carrying capacity under scrutiny, the applicability of the concept to communally-grazed areas in particular is also a much discussed topic (Ellis & Swift, 1988; Behnke & Scoones, 1992; Abel, 1993). The particular objective of keeping (rather than selling) livestock is a major point of discussion in this regard. Commercial livestock ranches aim to produce good quality and healthy livestock that can be sold on markets, whereas for many African communal pastoralists the main objective may be to have as many head of livestock as possible for social reasons and to reduce the risk of loosing all animals after a severe drought (Ellis & Swift, 1988; Behnke & Scoones, 1992; Bartels *et al.*, 1993; De Leeuw & Tothill, 1993). In the case where a pastoralist wants as many head of cattle possible, the same rangeland may have a higher carrying capacity (in the short term) than for the pastoralists who want strong and healthy animals.

Communal pastoralists often herd animals to specific grazing areas. These areas are the key grazing areas on which pastoralists rely for forage. The use of the key grazing areas is often bounded by traditional rules or family ties. Implementing a particular stocking rate based on a carrying capacity

is difficult in communal areas because social criteria in favour of high stocking levels may be more important to land owners then ecological arguments against high stocking rates (Bartels *et al.*, 1993; De Leeuw & Tothill, 1993).

Another confounding factor is that communal pastoralists often have multi-species herds. Calculating the carrying capacity for a multi-species herd involves a thorough investigation of the grazing or browsing habits of the animals and the amounts of feed they need. This makes calculating carrying capacity very difficult (De Leeuw & Tothill, 1993).

Carrying capacity is often perceived to be fixed (Stoddart, 1960) but due to the inherent variability of precipitation in semi-arid rangelands (Noy-Meir, 1973), rangeland production fluctuates constantly. Calculating a fixed carrying capacity for 10 years might therefore be unrealistic. Instead, carrying capacity varies from camp to camp and from year to year due variable rainfall.

The use of the carrying capacity concept is also limited to ecosystems with particular system dynamics. As described in previous sections, ecosystems can be divided into two types: equilibrium and non-equilibrium systems. Equilibrium systems develop towards a stable state under the influence of biotic factors, whereas non-equilibrium systems adapt to stochastic abiotic events such as rainfall. The density-dependent model, and therefore carrying capacity, is only applicable to equilibrium systems because the theory explicitly links vegetation change to herbivore density and vice versa (see: Ellis & Swift, 1988; Illius & O'Connor, 1999). Determining ecosystem dynamics is thus important to justifying the use of carrying capacity.

All in all, much doubt has been shed on whether the carrying capacity concept is useful for communal pastoralists or commercial pastoralists. Bartels *et al.* (1993) suggest that carrying capacity should not be used in communal areas. Instead, they advise that one uses the method proposed by Stoddart (1960). This method is based on continuous monitoring of the vegetation after an initial carrying capacity is determined; negative vegetation changes should be followed by reducing stocking density. This method is comparable to opportunistic management (Westoby *et al.*, 1989). Here, management interventions, such as reducing livestock density or controlled burning, are used to keep the vegetation in an optimal grazing state. For commercial livestock ranching, carrying capacity remains an important concept (see: Holechek *et al.*, 1995; Tainton, 1999). Use of carrying capacity for rangelands under commercial livestock ranching is less contentious for a number of reasons: Foraging habits can clearly be determined because often the ranches focus on cattle of a certain

variety. Ranches are often divided into grazing camps (=paddocks), which enables a manager to potentially determine the correct stocking rate of the particular camp. Game ranches have, however, similar problems to the communal livestock ranches. Many species graze the rangeland. Herds often keep to a particular vegetation type but can, in periods of food scarcity, use other areas that they normally do not use. This makes the practical implementation of stocking rates difficult, if not impossible.

4. Rangeland degradation and herbivory

Management-induced rangeland degradation is degradation caused by direct or indirect management actions that have negative effects on the rangeland ecosystem. Before discussing different management actions that can cause rangeland degradation and the ecosystem processes involved in degradation, the term *degradation* has to be defined. This section will also briefly introduce some different ways of detecting rangeland degradation.

4.1. When is change synonymous with degradation?

Degradation in arid and semi-arid areas has often been called desertification (Dregne, 1983; Mainguet, 1991; Arnalds, 2000). The term 'desertification' is, however, not clear and a generally acceptable definition of the term is still to be defined (Arnalds, 2000). Therefore, it might be clearer to use the term 'degradation' rather than 'desertification' (Walker & Noy-Meir, 1982). Different studies have proposed definitions for rangeland degradation. Within these different definitions, Archer & Stokes (2000) recognise the existence of two perspectives on rangeland degradation, namely the: socio-economic perspective and the ecological perspective. The socio-economic perspective of degradation mainly identifies degradation as a reduced carrying capacity for livestock, e.g. Abel & Blaikie (1989:113) state that rangeland degradation is the irreversible "...decline in the rate at which land yields livestock products under a given system of management". The ecological perspective, however, adds loss of biodiversity, primary production and nutrients to the definition of range degradation (Hoffman, 2000). In both socio-economic as well as ecological perspective, resilience is an important characteristic of a rangeland degradation definition (Archer & Stokes, 2000; Tongway & Hindley, 2000; Imeson & Cammeraat, 2002). Resilience is the capacity of a system to absorb disturbance (change) and restore soil and vegetation characteristics, such as nutrient levels and vegetation diversity, to the state it was in before the disturbance (Gunderson, 2000). Long-term degradation, and level of ecosystem functioning is difficult to detect by examining vegetation composition alone (Tongway & Hindley, 2000). Other factors, such as soil nutrient status, have to be incorporated to achieve this. The ecological perspective of degradation clearly puts more emphasis on these factors than the socio-economic perspective, and is therefore more suitable for a study such as this study.

4.2. Detecting rangeland degradation

4.2.1. Rangeland condition concept

In the previous section, it became clear that there are different views on rangeland degradation and that some studies solely use vegetation composition as an indicator of rangeland degradation and others take soil factors in consideration. Up to now, methods using vegetation composition only as an index of rangeland condition have been used extensively in South African agricultural research (Hoffman, 2000). Many of these methods are based on the same principles as the rangeland condition method presented by Dyksterhuis (1949). He suggested a method based on quantitative ecology that enables rangeland managers to determine the condition of their rangeland. Grouping of rangeland plants into three classes forms the basis of the theory. Plants are identified as (1) decreaser species; (2) increaser species; and (3) invader species. Decreaser species are plants that decrease in abundance under rangeland deterioration, increaser species increase under rangeland deterioration and invader species are plants that invade the rangeland with increasing grazing pressure (Dyksterhuis, 1949). The relative cover of the groups determines the rangeland condition. Figure 5 shows how rangeland condition deteriorates after 100 years of heavy grazing (often termed *overgrazing*). The figure also gives four condition categories (excellent-, good-, fair-, poor range

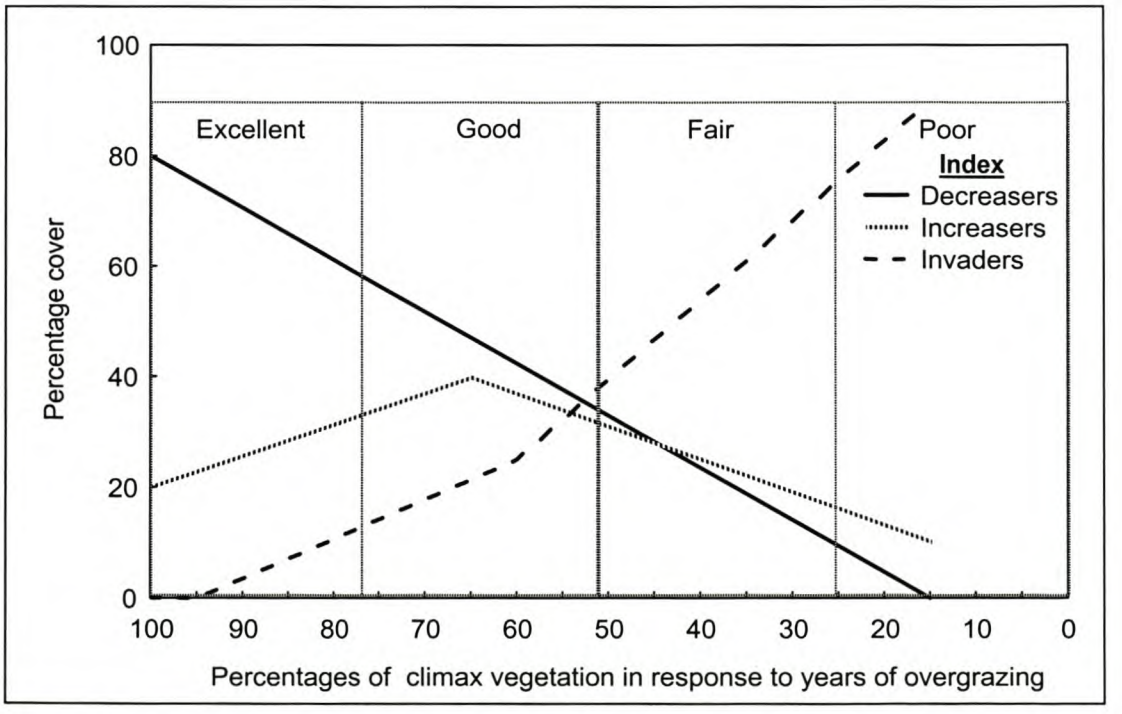

Figure 5: *Graph showing a quantitative basis for determining range condition* (Dyksterhuis, 1949)

condition) with the corresponding vegetation cover of the three plant types (decreaser, increaser and invader).

4.2.2. Soil nutrients and degradation detection

Ward *et al.* (1998) used soil nutrients, as well as vegetation composition, as an indicator for rangeland degradation, as do many other ecological studies (e.g. Tolsma *et al.*, 1987; Moleele & Perkins, 1998; Dougill *et al.*, 1999; Imeson & Cammeraat, 2000; Fernandez-Gimenez & Allen-Diaz, 2001; Krogh *et al.*, 2002). As mentioned earlier, vegetation composition and/or cover alone do not always give a good indication of ecosystem processes and long-term degradation. Vegetation may be influenced by seasonal variability in rain or the occurrence of long droughts. The soil nutrients nitrogen (N) and phosphorous (P) are, after water, major limiting resources for rangeland ecosystems (Cole, 1982; Berg *et al.*, 1997; Snyman, 2002). When these nutrients are lost from the system, then the vegetation will not completely recover. These parameters should therefore be included if rangeland condition is to be assessed. Which vegetation and soil parameters are often used to detect rangeland degradation will be discussed in later sections.

4.2.3. Piospheres (grazing gradients)

Lange (1969) described sheep track densities around water-points, and called the phenomenon of having different levels of track densities around water-points 'piospheres' (derived from the Greek word *pios* (to drink)). The density of tracks was high near to the water-point and reduced as one moves away from the water-point. As track density is positively correlated with grazing intensity (piospheres are therefore also called 'grazing gradients'), piospheres are useful to record changes in rangeland vegetation for management purposes (Lange, 1969). This method has since been used by many other studies to study the impact of herbivores on rangeland vegetation and to identify rangeland degradation (e.g. Tolsma *et al.*, 1987; Andrew, 1988; Pickup & Chewings, 1994; Jeltsch *et al.*, 1997; Moleele & Perkins, 1998; Thrash, 1998; James *et al.*, 1999; Makhabu *et al.*, 2002). Studying piospheres has proved to be particularly useful in semi-arid and arid ecosystems because these systems have very high spatial variability (patchiness) of vegetation and soils, which makes it difficult to assess where best to measure the effects of herbivory. Studying piospheres is effective in such systems because the gradient in grazing intensity is known, facilitating the separation of effects caused by herbivory and inherent spatial variability.

4.2.4. Remote sensing

Tomlinson *et al.* (2002) examined the smallest economically-viable ranch size for three management systems (commercial cattle ranching, communal livestock ranching, and game ranching). The study found that commercial cattle ranches in semi-arid rangelands (annual precipitation between 600 and 800 mm) need to have at least 300 ha of rangeland in order to make a profit, whereas commercial game ranches even need a minimum of 1 000 ha to be able to make a profit. Because ranches are so large, there is a need for rangeland evaluation methods that are able to survey large areas at minimal costs. In these areas, remote sensing can be a more rapid and economically-viable alternative than ground-based survey techniques (DeSoyza *et al.*, 2000). Drawbacks of remote sensing are the low spatial resolution (although high spatial resolution satellite images become available (Space Imaging, 2003)), and the inability to distinguish among plant species (DeSoyza *et al.*, 2000). However, more and more arid and semi-arid rangeland ecology studies incorporate remote-sensing techniques in their studies (e.g. Stafford Smith & Pickup, 1993; DeSoyza *et al.*, 2000; Kepner *et al.*, 2000; Pickup *et al.*, 2000; Kumar *et al.*, 2002). Several studies examining piospheres have also used remote-sensing techniques to detect degradation around water-points. These studies examined livestock track patterns around water-points using aerial photographs (Lange, 1969), while more recent studies use satellite images to detect changes in vegetation cover along piospheres (Bastin *et al.*, 1993; Pickup, 1994; Pickup & Chewings, 1994; Saltz *et al.*, 1999; Ludwig *et al.*, 2000).

5. Vegetation and herbivory

Rangeland degradation can be identified by means of a variety of parameters. Soil and vegetation are two important parameters that are affected by degradation. This section discusses how rangeland vegetation is affected by herbivory.

5.1. Biodiversity

Several studies have found reduced biodiversity under heavy grazing (West, 1993; Milton *et al.*, 1994; Ritchie & Olff, 1999) although some report that herbivores have relatively little effect on rangeland biodiversity (Ward, in press). Other studies have reported an increased diversity of annual species and a lowered diversity of perennial species for areas with high grazing pressure (Todd & Hoffman, 1999). Biodiversity is an important measure of degradation - ecologically and socio-economically it holds much value. West (1993) lists four basic reasons that demonstrate its importance:

1. People feel a moral obligation to protect species from extinction.
2. There is an aesthetic reason to conserve a multitude of species (e.g. gardens, zoos, nature reserves).
3. It is economically beneficial to be able to conserve a myriad of species for possible future commercial use (e.g. medicine, food).
4. A high biodiversity ensures that a wide array of services are provided by natural ecosystems (e.g. climate, soil fertility, etc.).

A decline in biodiversity could therefore cause degradation of the ecosystem.

5.2. Soil change and species composition

As mentioned earlier, the species composition of an ecosystem is another important indicator of rangeland condition. Species composition of a site is determined by the species present and their percentage cover in a particular area. There are several ways through which herbivores may affect species composition, of which a change in soil nutrients is often an important cause of the change.

Herbivory often brings a change in soil nutrients and characteristics. Variations in soil nutrients can affect species composition. For example, Tolsma *et al.* (1987) and Moleele & Perkins (1998) found an

increase in woody encroachers, such as various *Acacia* species, around water-points. Tolsma *et al.* (1987) suggest that the increase in woody species is caused by seeds in cattle dung deposited near water-points where soil is enriched with nutrients, and therefore have a high germination and survival rate. It is not only woody species that benefit from a change in soil nutrients due to herbivory. Svejcar & Sheley (2001) remarked that "... widespread heavy grazing during the early part of this century may have predisposed [North American rangeland] communities to conversion from perennial to annual dominance". The reason is that increased nitrogen gave the invading annual grass species a competitive advantage over the native North American perennial grasses. Similar effects were found by McLendon & Redente (1991).

5.3. Vegetation cover

Vegetation cover is an important characteristic of rangelands, as it directly and indirectly affects the secondary productivity of a rangeland. When vegetation composition and nutrient value of the vegetation remain optimal, it may be clear that an increase in vegetation cover means an increase in the ability to feed an increasing number of herbivores (Odum, 1971; Holechek *et al.*, 1995; Snyman, 1998; Owen-Smith, 1999; Snyman, 1999). Here, vegetation cover directly affects secondary productivity. Indirectly, vegetation cover positively affects certain processes that create better conditions for plant growth. Vegetation cover increases the retention of water after rainfall and decreases runoff that can cause serious erosion and loss of nutrients (Snyman, 1998; Whitford *et al.*, 1998; De Soyza *et al.*, 2000). Herbivores affect vegetation cover through trampling and selective grazing (Tolsma *et al.*, 1987; Andrew, 1988; Ward, 2003). Vegetation cover is, therefore, important when evaluating rangeland condition.

Increasingly, studies make use of remote-sensing techniques to determine rangeland condition. These studies often determine rangeland condition on the basis of vegetation cover (e.g. Bastin *et al.*, 1993; Pickup, 1994; Pickup & Chewings, 1994; Saltz *et al.*, 1999; Ludwig *et al.*, 2000). An estimate of vegetation cover is often derived from simple aggregative indices such as the normalised difference vegetation index (NDVI). This index is calculated using the following formula (in which IR = infrared band, and R = red band):

$$NDVI = (IR-R)/(IR+R) \text{ (Campbell, 2002)}$$

However, this index has been criticised by Saltz *et al.* (1999), who point out that in hyper-arid environments, vegetation cover (as measured by NDVI) is less useful due to the inherently high percentage of bare soil which may have similar reflective values to certain vegetation types. Pickup *et al.* (1993) had also identified this as a problem when determining vegetation cover in Australian rangelands using NDVI. They developed a vegetation cover index, named PD54, using the visible green and visible red bands from Landsat MSS data (Pickup *et al.*, 1993). This vegetation index is "… relatively insensitive to differences in vegetation greenness which most others indices are not" (Pickup & Chewings, 1994:599).

In addition to remote sensing one can also measure vegetation cover in the field. Here measurements are more accurate. However, the method is more time consuming, can only be done for relatively small areas, and is therefore costly. De Soyza *et al.* (2000) use bare patch size instead of vegetation cover. De Soyza *et al.* (2000) describe a method to measure bare patch size, together with other indicators of rangeland health.

6. Soils and herbivory

Soil condition is determined by the physical structure, chemical composition, nutrient content, and biological processes in the soil (Sims, 2000). Soil condition is an important measure to add to rangeland condition studies because it gives a better indication of long-term degradation of a site (e.g. Dougill *et al.*, 1999; Turner, 1999). Vegetation composition is often influenced by short-term drought stress, or defoliation. This does not necessarily indicate inter-annual rangeland degradation because plants may recover following good rainfall years (Turner, 1999). Degradation of the soil, however, does indicate long-term problems for productivity and biodiversity as the capacity of the environment is thereby reduced.

The following section deals with soil parameters such as key nutrients, components and indicators of soil condition. It also discusses how livestock and game may affect soil condition through defecation, urination and trampling. Through understanding the interactions between herbivores and soil condition, we can link changes in soil parameters to certain management systems and related activities.

6.1. Soil characteristics

In this section, organic carbon, pH and electrical conductivity will be discussed in relation to soil condition. These three indicators are reliable and easy to measure and therefore useful indicators for a soil degradation survey.

6.1.1. Organic matter and soil fertility

Plants and other organisms do not rely on soil organic carbon (C) as their supplier of C; nearly all the C they need is derived directly from the atmosphere. Nevertheless, soil organic C is of interest from the viewpoint of soil fertility and soil physical properties (Stewart *et al.*, 1987; Berg *et al.*, 1997; Snyman, 1999). Organic C contains complex compounds that include nutrients such as nitrogen (N) and phosphorus (P), among many others. After decomposition, these nutrients become available to plants. Studies have found positive correlations between organic C and N and P concentrations in the soil (Turner, 1998; Ward *et al.*, 1998; and to a lesser extent Hiernaux *et al.*, 1999). Moreover, soil organic C strongly influences the cation exchange capacity (CEC) of a soil (Rowell, 1994); as organic

matter increases, so does the CEC. Increasing CEC increases the capacity of the soil to retain nutrients and avoid leaching of nutrients (Sims, 2000). Furthermore, soil organic C increases the stability of soil aggregates, which make clay soils less sticky and thus increases the soil penetrability for roots, increases soil aeration and makes sandy soils more cohesive (Janssen *et al.*, 1990; Snyman, 1998). Of greater importance to ecosystems where water availability is the limiting factor for plant growth, such as arid and semi-arid ecosystems, is the water retention capacity of organic C. Per gram of organic C, the water retention capacity is increased by 1-8 cm^3 (Janssen *et al.*, 1990). Many rangeland degradation studies have recorded organic C as an indicator of grazing impact (e.g. Tolsma *et al.*, 1987; Moleele & Perkins, 1998; Ward *et al.*, 1998; Dougill *et al.*, 1999; Imeson & Cammeraat, 2000; Fernandez-Gimenez & Allen-Diaz, 2001; Krogh *et al.*, 2002). In sum, organic C content is a crucial measure of soil fertility and soil structure and should be included in any soil quality study.

6.1.2. pH and degradation

Nitrification occurs when ammonium (NH_4^+) is converted to nitrate (NO_3^-). During this process 2 H^+ ions are released. This process has an acidifying effect on the soil. Nitrification occurs in places where there is a high availability of NH_4^+ that can be converted to NO_3^-. Deposition of N in urea can actually have the reverse effect on soil pH. Urea contains ammonia (NH_3), which is converted to ammonium (NH_4^+) when deposited. After deposition the following reaction occurs binding H^+ and increasing soil pH:

$$NH_3\,(aq) + H^+ \rightarrow NH_4^+ \text{ (Stumm \& Morgan, 1996).}$$

There is a strong increase in pH after this reaction in the vicinity of the deposition (Killham, 1994). However, the pH increase is often short lived as nitrification quickly sets in (Bloom, 2000). A temporarily higher soil pH may therefore be due to urea deposition, but a low soil pH may also indicate urea deposition, depending on the time that the urea was deposited (Bloom, 2000).

In the case of salinization, however, soils can become more alkaline when sodium (Na) reacts with CO_3^{2-}. In soils containing Na, the following reaction takes place:

$$2\,Na^+ + CO_3^{2-} + H_2O \rightarrow 2\,Na^+ + HCO_3^- + OH^- \text{ (Bloom, 2000).}$$

Salinization of soils has been attributed to desertification and degradation of arid and semi-arid ecosystems around the world (Mainguet, 1991; Dregne, 1983), as plants have more difficulty taking up

water to the point that they may die (Sims, 2000). In rangelands, salt supplements for cattle may cause localised salinization due to spillage of salt.

The process of acidification or, on the other hand, soils becoming more alkaline, can affect plant growth negatively. pH is therefore a useful indicator of degradation (Bell, 1982; Holechek *et al.*, 1995) and is relatively easy to acquire. Soil pH influences many different plant nutrient-related processes (Bloom, 2000). The most important are reduced P availability in alkaline soils (Whitehead, 2000), reduced N fixation of legumes under acid soil conditions (Coventry & Evans, 1989; Graham & Vance, 2000) increased N availability after increased pH (Curtin *et al.*, 1998). Soil pH also provides information on possible toxic levels of aluminium and manganese in acid soils under certain circumstances (Ritchie, 1989; Sims, 2000) and increased pH due to salinization (Postiglione, 2002).

The processes mentioned in this section do not necessarily lead to a change in soil pH, as the soil is able to buffer a certain amount of H^+. Some soils have high concentrations of free calcium carbonate ($CaCO_3$). The reaction that follows when $CaCO_3$ reacts with H^+ is:

$$CaCO_3 + 2\ H^+ \rightarrow\ Ca^{2+} + CO_2 + H_2O \text{ (Bloom, 2000).}$$

In this case, $CaCO_3$ buffers the free H^+ so there is no further decrease in pH (Rowell, 1994). Soil organic C too has such a pH buffering capacity. This is caused by compounds such as benzoic acid (Ar-COOH) and phenol (Ar-OH) that can be found in complex soil organic C compounds (Bloom, 2000). The more soil organic C there is, the more H^+ that can be buffered. All-in-all, soil pH gives an indication of nutrient availability and possible aluminium or manganese toxicity (Sims, 2000) and is therefore a useful and easy measure of soil fertility.

6.1.3. Conductivity and desertification

Soil electrical conductivity (EC) is a measure of the concentration of ions present and their electrical charge (Rowell, 1994) and is often used to measure salt content of soils. Salinity is an important factor in soil fertility as it reduces soil quality and productivity (Hao & Chang, 2003). EC can be used as an indicator of desertification (Dregne, 1983) and is mostly used to monitor irrigated lands as they are prone to salinization when brackish irrigation water is used (Dregne, 1983; Mainguet, 1991). Soil EC can also be used to monitor the effects of livestock on soil condition. Hao & Chang (2003) found a significant increase of soil EC over a period of 25 years of cattle manure application to a semi-arid cropland. They point out that manure often has a high salt content due to the supplementary feed given to cattle, which contains high concentrations of salt.

6.2. Soil nutrients

6.2.1. Nutrient gradients

A number of studies have focused on the effect of grazing gradients on soil nutrients (e.g. Tolsma *et al.*, 1987; Manley *et al.*, 1995; Turner, 1998; Schuman *et al.*, 1999). There are two main causes of nutrient change due to herbivore density: (1) nutrient transport to a water-point by herbivores, and (2) nutrient increase due to vegetation change. The transport of nutrients to central water-points and pans (also called *centripetal* movement of nutrients) has been described by a number of studies (e.g. Tolsma *et al.*, 1987; Moleele & Perkins, 1998; Turner, 1998; Fernandez-Gimenez & Allen-Diaz, 2001). Tolsma *et al.* (1987) found that within 20 m around a water-point, phosphorus levels increased so dramatically that it could be compared to fertilisation of 55 kg P ha^{-1} $year^{-1}$. The phosphorus increase observed near the water-point, however, may have caused a 17% reduction in P in the actual grazing area around the water-point. Turner (1998) found similar phosphorus accumulation around water-points but did not find a nitrogen gradient. He notes that nitrogen is less likely to build up near to water-points due to NH^{4+} volatilisation and leaching. Vegetation change due to herbivore density can also affect soil nutrients (Manley *et al.*, 1995; Schuman *et al.*, 1999). Manley *et al.* (1995) found that soil organic carbon (org. C) increased under heavy grazing in comparison with non-grazed exclosures. They note that this was partly caused by an altered plant composition. The abundance of plants with a higher root:shoot ratio increased, and the distribution of C above ground and below ground changed. These effects may be negative for herbivores as Moleele & Perkins (1998) note. They found that woody unpalatable plant species flourished on sites where there was increased soil N. All-in-all, movement of soil nutrients to a central water-point slowly degrades the rangeland as production of palatable grasses is, in the long run, reduced due to nutrient depletion and unpalatable shrubs dominate around the water-points.

6.2.2. Nitrogen

Nitrogen (N) is the most important building block for organisms (Scholes & Walker, 1993), is essential to plants (Whitehead, 2000), is the principal cause of ecological productivity, and is therefore important for soil degradation assessments (Berg *et al.*, 1997; Dougill *et al.*, 1999). The following section will discuss the sources of N, soil processes that make N available to plants, and how N is lost from the soil.

The N-cycle has a number of pools from which plants can draw N directly or indirectly. These are: soil organic carbon (C), atmosphere, and soil parent material. Nearly 95% of the total N found in soils occurs in soil organic C. This N is not directly available to plants: microbes have to decompose the organic C first, after which the N will be available to plants (Whitehead, 2000). The atmosphere is also a potential supplier of N. Atmospheric N (N_2) can be deposited through airborne dust or with precipitation (Scholes & Walker, 1993). Another way that the atmosphere supplies N to plants is indirectly. Some legumes have a symbiotic relationship with rhizobial bacteria (Scholes & Walker, 1993; Killham, 1994). In this relationship, the plant provides energy to the bacteria and the bacteria fix N_2 from the atmosphere that is used by the plant. Soil parent material only supplies a negligible amount of N to the soil (Blackmer, 2000). Plant tissue harbours a lot of N that becomes available to other plants when the plant tissue is eaten by herbivores or when the plant dies (Blackmer, 2000). In the case of plants eaten by herbivores, most of the N is returned to the soil via urine and dung.

Two N compounds are available to plants: ammonium (NH_4^+) and nitrate (NO_3^-). Ammonium and nitrate become available in the soil through the processes of ammonification and nitrification. Ammonification occurs when soil organic N in organic matter is converted to NH_4^+ by soil microbes (Norton, 2000). In this process, the C:N ratio of the organic matter that is decomposed determines how much NH_4^+ will become available to plants. Soil microbes that decompose organic N use a portion of the produced NH_4^+. In the case that more NH_4^+ is produced than is utilised by soil microbes, we speak of N mineralisation (Norton, 2000). In the case that all the NH_4^+ produced during decomposition is used by the soil microbes, the term *N immobilisation* is used (Norton, 2000). The C:N ratio is an indication of the amount of NH_4^+ that will potentially be available to plants. The lower the C:N ratio, the higher the NH_4^+ potentially available to plants and *vice versa*. C:N ratios that enable N mineralisation or immobilisation vary. Scholes & Walker (1993) found that an organic substrate with a C:N ratio < 11.5:1 caused N mineralisation, whereas a C:N ratio > 16:1 caused immobilisation. Whitehead (2000) gives the following ratios: C:N ratios $\leq$ 20:1 cause N mineralisation, and C:N ratios $\geq$ 30:1 cause N immobilisation. The C:N ratios of excreted urea and dung differ as dung contains a higher percentage of organic matter. For cattle dung, the C:N ratio is 20:1 and for urine it is 3.9:1 (Whitehead, 2000). The low C:N value of cattle urine indicates that a large portion of the N available in the urine will be mineralised. Dung has a higher C:N ratio; less N will therefore be mineralised as

most of the N will be utilised by soil microbes (Whitehead, 2000). Deposition of urine therefore increases plant available N, whereas dung affects soil N much less.

Most of the ammonium (NH_4^+) produced during ammonification is converted into nitrate (NO_3^-) by bacteria; this process is called nitrification (Norton, 2000). The nitrification process takes two steps to complete. Both transformations are caused by activity of *Nitrosomonas* and *Nitrobacter* bacteria respectively (Norton, 2000). The two reactions are as follows:

$$2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 4H^+ + 2H_2O + e^- \text{ (Killham, 1994).}$$

$$N0_2^- + H_2O \rightarrow NO_3^- + 2H^+ + 2e^- \text{ (Norton, 2000).}$$

Unlike NH_4^+, NO_3^- cannot bind itself to clay as it is negatively charged. Due to its negative charge, NO_3^- can move freely and therefore is readily available to plants. On the other hand, it is therefore also susceptible to leaching and denitrification.

Nitrogen can be lost from the nitrogen cycle through different ways: volatilisation, leaching, denitrification and nitrification of N. In semi-arid rangelands, nitrification is not a major cause of N loss. This is mainly due to the low precipitation. Due to the low precipitation, leaching of NO_3^- is not likely to be widespread (Scholes & Walker, 1993). Denitrification of NO_3^- will neither occur often as this process needs anaerobic conditions to convert NO_3^- into gaseous compounds such as NO, N_2O and N_2. Volatilisation, however, is an important cause of N loss (Scholes & Walker, 1993; Whitehead, 2000). Volatilisation is a process where the gaseous ammonia (NH_3) is formed and escapes to the atmosphere. This process occurs during ammonification, nitrification and with the deposition of urine and, to a lesser extent, dung. When urine or dung is excreted, available NH_4^+ in the soil reacts with N in the urine of animals, creating NH_3 that is lost to the atmosphere (Scholes & Walker, 1993; Whitehead, 2000). Whitehead (2000) adds that volatilisation increases with increasing temperature and is greatest from soils of high pH and low cation exchange capacity. Fire can also cause the loss of nitrogen due to volatilisation (Scholes & Walker, 1993).

6.2.3. Phosphorus

After water and nitrogen (N), phosphorus (P) is the limiting nutrient for plant growth (Holechek *et al.*, 1995), and is a key element to any organism (Scholes & Walker, 1993). P in the soil mainly originates from the parent material. Unlike the N-cycle, the atmospheric pool of the P-cycle plays a negligible role (Scholes & Walker, 1993; Whitehead, 2000). Soil P occurs in an inorganic form and an

organic form. The sizes of these pools varies per soil and depend largely on the soil parent material (Sharpley, 2000). In arid soils the organic pool is an important P-pool in the P-cycle (Scholes & Walker, 1993; Sharpley, 2000). Organic P, however, is not directly available to plants. It first has to be mineralised by soil microbes as is the case with organic N. C:P ratios are, like C:N ratios for N mineralisation, an indication whether P is potentially mineralised. C:P ratios $\geq$ 200:1 cause immobilisation whereas C:P ratios < 200:1 most probably cause mineralisation of P (Whitehead, 2000). However, these values vary from soil to soil. Plant tissue harbours another P-pool in the P-cycle. This P is returned to the soil when the plant dies or when it is eaten. Dung from animals contains 95% of the P excreted by herbivores (Whitehead, 2000). Urine only contains a negligible amount of P (Whitehead, 2000).

Phosphorus is available to organisms in the form of phosphate ions ($H_2PO_4^-$ and HPO_4^{2-}). The ratio between the two ions in the soil depends on the pH of the soil. In a more acidic soil, $H_2PO_4^-$ is dominant; up to 95% of phosphate ions are $H_2PO_4^-$ (Whitehead, 2000). In alkaline soils, however, higher concentrations of HPO_4^{2-} can be found and constitute up to 45% of the total phosphate ions (Whitehead, 2000).

Leaching of phosphorus (P) is negligible. The phosphate ions are quickly absorbed by the soil. Volatilisation of phosphorus (P) does not occur in high quantities as does nitrogen; surface runoff has a bigger impact on phosphorus loss (Whitehead, 2000). Surface runoff, however, can cause P loss (Whitehead, 2000).

6.3. Soil micro-organisms

Soil micro-organisms play an important role in soil nutrient dynamics and are a large source of plant-available nutrients (Singh *et al.*, 1989). As has been discussed in previous sections, soil micro-organisms help break-down complex organic carbon (C) molecules, after which nutrients become available to plants. A reduction in these micro-organisms can negatively affect the ecosystem as fewer nutrients become available to break-down organic C. Holt (1997) studied the effect of heavy grazing on the biomass of soil micro-organisms. This study found that the biomass of soil micro-organisms was consistently lower in heavily-grazed plots compared to plots that were lightly grazed. The reduction was attributed to "… a combination of lower soil moisture and a lower rate of organic matter input as a result of a reduction in herbage biomass" (Holt, 1997:147). Soil micro-organisms play a vital role in rangeland ecosystems. A reduction in soil micro-organism biomass can cause a

change in the availability of soil nutrients. Monitoring the biomass of soil micro-organisms can therefore give a good indication of soil condition.

Measurement of a soil's microbial biomass can be done using different methods. Mostly, measurement is indirect via the measurement of CO_2 produced by the soil micro-organisms as a measure of their total mass. Other measurements are based on microscope counts (for methods, see: Jenkinson & Ladd, 1981; Horwath & Paul, 1994). Comparison of the results of different tests show that results vary among tests (Wardle & Ghani, 1995). This is partly because the different measurements do not necessarily measure the same thing. Some measure CO_2 production whilst others convert the result directly into biomass (Wardle & Ghani, 1995).

7. This study

7.1. Objectives

The main objective of the study is to compare the negative impact (degradation) of three different rangeland management systems (i.e. commercial livestock ranching, communal livestock ranching, and game ranching) on a semi-arid rangeland ecosystem near Barkly West, Northern Cape, South Africa. Abiotic factors, such as precipitation and temperature, are not easily influenced by management systems. In this study we therefore want to focus on degradation caused by biotic factors only. The climate of the study site is very variable, with a high spatial and quantitative variability of the precipitation (for Barkly West the mean annual rainfall = 388 mm (C.V. = 39 %) (Kraaij, 2002)). This ecosystem could therefore potentially be classified as a non-equilibrium system. However, we predict that the management systems affect the ecosystem in spite of the variable nature of the ecosystem. We use gradients of grazing intensity (*piospheres*) in order to differentiate degradation induced by abiotic forces from degradation caused by biotic forces. This allows us to correlate indicators of degradation to grazing intensity. We use two sets of degradation indicators:

1. vegetation composition, and density (see *Vegetation and herbivory*);
2. soil chemical and physical characteristics (see *Soils and herbivory*).

7.2. Study site description

The study location was chosen because of the close proximity of the three management systems (commercial livestock ranching, communal livestock ranching, and commercial game ranching). Also, climatic conditions and soils were similar among management systems, enabling a comparison between the different management systems.

7.2.1. Location and climate

The study site is located west of Kimberley, Northern Cape province, South Africa. The co-ordinates of the approximate centre of the study site are 28°36' S, 24°28' E (1 125 m a.s.l.) (Figure 6). Measurements were conducted on four ranches: (1) Pniel Estates, (2) Barkly West commons, (3) Nooitgedacht, and (4) Rooipoort. The focus area was Pniel Estates, owned by the Evangelical Lutheran Church of South Africa. It consists of a 3,000 ha communal livestock ranch, 12,000 ha

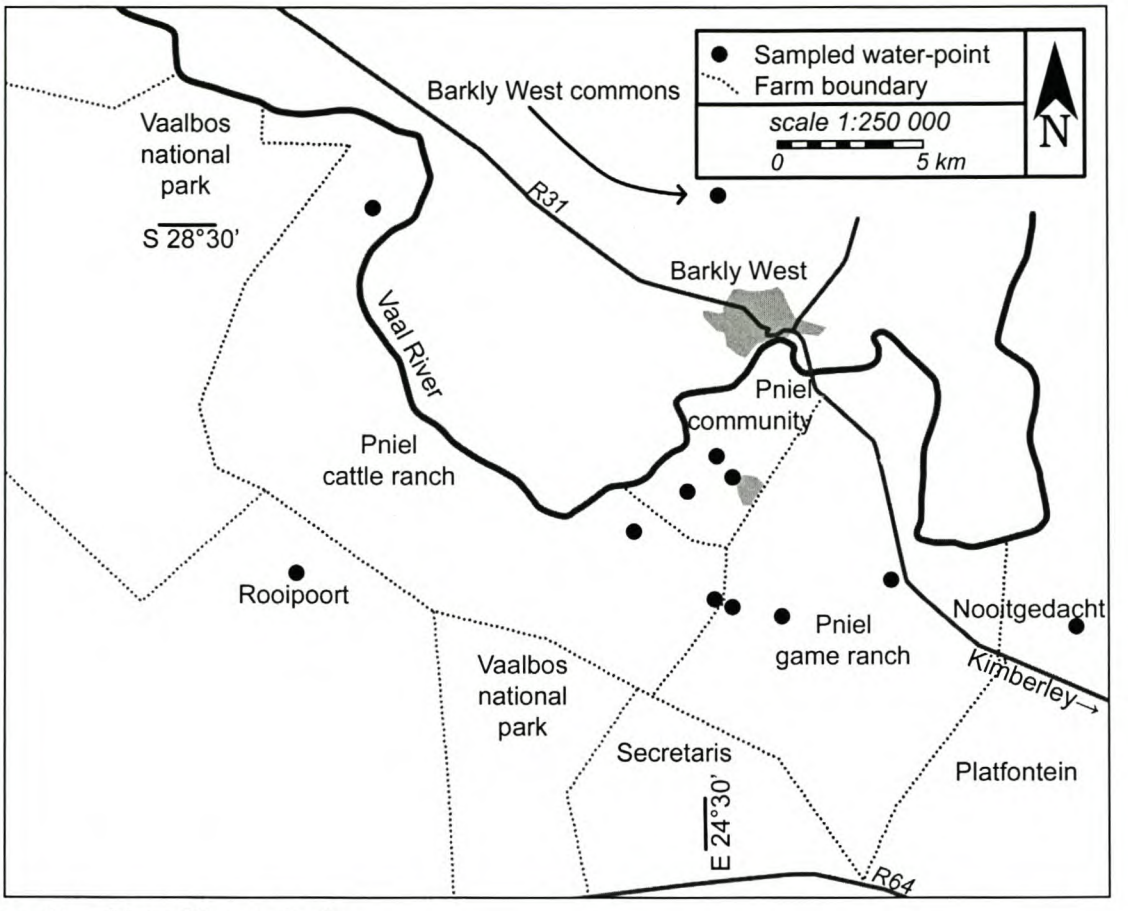

Figure 6: *Map of the study site.*

commercial livestock ranch, and a 10,000 ha game ranch. The Barkly West commons, Nooitgedacht and Rooipoort ranches are communal livestock -, commercial cattle - and game ranches respectively and served as replicates for the Pniel Estates' measurements. The study site's climate can be described as semi-arid with a mean annual precipitation of 388 mm (C.V. = 39 %) (Kraaij, 2002) at Barkly West (which borders Pniel Estates), and 414 mm (South African Weather Service, 2003) for Kimberley (30 km east of Pniel Estates). Rains are concentrated in the summer between November and April. Mean annual maximum temperature is 26°C and the mean annual minimum temperature is 11°C. Maximum air temperatures can rise well above 30°C and nights can occasionally experience frost (South African Weather Service, 2003).

7.2.2. Geology and Soil

The study site is located in the Kalahari basin which is, together with the Congo basin, one of the oldest basins in Africa - it was uplifted more than 1500 million years ago (Taylor, 1996). The geology of the study site consists of 5 main formations:

1. the oldest deposit is the Ventersdorp andesite (lava) formation which was deposited about 2700 million years ago during the Precambrian era;
2. a formation deposited during the Precambrian era some 2100 million years ago, and consisting of white quartzite;
3. Dwyka shales and tillites deposited during the Palaeozoic era or Karoo period and are 280 million years old together with dolerite and basalt pipes that were formed 180 million years ago;
4. calcrete formations formed in the Tertiary era 20 million years ago; and
5. Hutton sands, were formed recently (pers. comm. L. de Meillon[1], 2002).

Not all of these five formations are important to the study. The majority of the surface is covered by andesite, calcrete or Hutton sands. The other two formations, Dwyka shales and tillites, and white quartzite, only surface in isolated patches along the steep banks of the Vaal River.

Soils that can be found in the study site can largely be grouped into clayey -, sandy -, and calcareous soils, belonging to the andesite -, Hutton sands -,and calcrete formations respectively. Clayey soils are derived from the Ventersdorp andesite with variable amounts of sand and gravel (pers. comm. L. de Meillon, 2002). The clayey soils are found on and along the andesite outcrops along the Vaal River. Sandy soils can be up to 8 m deep and consist of 95% quartz (Van Riet and Louw & Metaplan, 1999). Most of the sand derives from the Kalahari and some from local weathered rock (pers. comm. L. de Meillon, 2002).

7.2.3. Vegetation classification

The study area is a rangeland. Blench & Sommer (1999:7) define rangelands as "... geographical regions dominated by grass-like species with or without scattered woody plants." Another definition that can be used to define rangeland is the one used by Holechek *et al.* (1995:1) who state that a rangeland is "... uncultivated land that will provide the necessities of life for grazing and browsing animals." These two definitions place emphasis on different aspects of rangelands; Blench & Sommer (1999) emphasise the vegetation composition of sites that may be classified as rangelands, whereas Holechek *et al.* (1995) (and others e.g. Stoddardt *et al.*, 1975 and Allen, R.E., 1990) are, however,

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more concerned with the agricultural use and value of these ecosystems to differentiate them with other ecosystems. Not only is the term 'rangeland' defined in many different ways, placing emphasis on different aspects of vegetation or use, these definitions often also include many different ecosystems. They may include grasslands, desert shrublands, savanna woodlands, forests and tundra (Holechek *et al.*, 1995). The term 'rangeland' therefore might not be useful to clearly delimit a particular vegetation type. In this article, the term 'rangeland' will therefore be used to refer to the study site but not to a particular vegetation type.

To be more specific and use a term that describes the vegetation in more detail one has to use another term. Two rangeland types can be found in the study site: savannas and shrublands. A savanna is, "... a tropical vegetation type in which ecological processes, such as primary production, hydrology and nutrient cycling, are strongly influenced by both woody plants and grasses, and only weakly influenced by plants of other growth forms" (Scholes & Walker, 1993:4). Walter (1971) mentions that savannas are found in the tropical summer-rain zone with a dry period of 2.5 - 7.5 months, while others emphasise the dominance of C_4 grasses (Huntley, 1982). Solbrig *et al.* (1996) simply define savannas as "... a continuous layer of graminoids (grasses and sedges) with a discontinuous layer of trees and/or shrubs." The one factor that nearly all savanna definitions have in common is that grasses and woody plants play an intrinsic role in all savanna ecosystem processes and also that there are distinct dry and wet seasons. In the savanna of the study site, tree species *Acacia erioloba* and *A. tortilis* and grasses species as *Schmidtia pappophoroides* and *Eragrostis lehmanniana* are dominant (Kraaij, 2002). The other rangeland type that can be found in the study site are shrublands, where shrubs less than 2 - 3 m dominate (Holechek *et al.*, 1995; Scholes & Walker, 1993). *Acacia mellifera* and *A. tortilis* are the dominant shrub species in this vegetation type and *Eragrostis lehmanniana* and *Aristida congesta* the dominant grass species. In South Africa, the term 'Kalahari thornveld' proposed by Acocks (1988) is more commonly used to characterise this vegetation. He describes this vegetation as *Acacia erioloba* savanna and *Acacia* shrubland. The more recent classification of Low & Rebelo (1998) classifies the vegetation as belonging to the savanna biome, and being part of the 'Kimberley thorn bushveld'.

7.2.4. Management systems in the study site

The study site is divided into areas that are managed using one of the following three management systems: commercial livestock ranching, communal livestock ranching, and game ranching.

Pniel Estates includes a 12 000 ha commercial livestock ranch. The commercial livestock ranching is mainly characterised by single species herds (in this case cattle) that graze camps for a certain period before moving to a next camp. The ranch has four cattle herds, each about with about 180 head (varying somewhat from year to year). The total number of cattle varies around 750 head of cattle (Department of Agriculture – Veterinary Services[2], 2002). This equals to about 750 Large Stock Units (LSU), as one LSU is defined as a bovine animal of 450 kg, increasing in mass by 500 g daily on forage with a digestibility of 55 % (Meissner *et al.*, 1983). The ranch uses a rotational grazing system whereby the herd rotates to another camp when a grazing camp does not have enough fodder (determined visually, approximately every 90 days). Each grazing camp has access to a central water-point where the animals also receive supplementary nutrients and salts. The cattle ranch produces calves for meat production, and when old enough the calves are sold to feedlots.

On one section of Pniel Estates, a poor black community of about 150 families occupies about 3,000 ha, where they keep livestock such as sheep, goats, donkeys, horses and cattle. About 150 sheep, 120 goats, 150 cattle, 85 horses, and 40 donkeys are kept on the communal livestock ranch (= 320 LSU) (Department of Agriculture – Veterinary Services, 2002). The community applies a multi-species continuous grazing system. The herd grazes the entire area throughout the year. Some animals are herded by their owners but a coordinated grazing system is not present. The animals do use seasonal water-points, but generally go to the Vaal River to drink. Few livestock owners feed their animals extra nutrients or salts. Animals are used for a multitude of purposes among which, traction, dairy products, meat, and social exchange are important uses.

Pniel Estates have, adjacent to the commercial livestock ranch (Figure 6), a commercial game ranch. The game ranch has many different species of wildlife (912 LSU)(in this case, white rhino (*Ceratotherium simum*), buffalo (*Syncerus caffer*), giraffe (*Giraffa camelopardalis*), eland (*Taurotragus oryx*), kudu (*Tragelaphus strepsiceros*), red hartebeest (*Alcelaphus buselaphus*), waterbuck (*Kobus ellipsiprymnus*), sable antelopes (*Hippotragus niger*), blue wildebeest (*Connochaetes taurinus*), springbok (*Antidorcas marsupialis*), warthogs (*Phacochoerus aethiopicus*), steenbok (*Raphicerus campestris*), among many others). These different species roam over the entire 8,000 ha but mainly

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restrict themselves to certain vegetation types within the camp. Water-points are scattered throughout the commercial game ranch where animals are also given salt supplements. The animals are used for trophy hunting, venison hunting, and ecotourism.

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A comparison of the effects of different rangeland management systems on plant species composition, diversity and vegetation structure using grazing gradients around water-points in a semi-arid savanna

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Abstract

Most of South Africa's land surface is arid or semi-arid rangeland. Three management systems exploit these areas: commercial livestock ranching, communal livestock ranching, and game ranching. The way in which these management systems affect rangeland ecology is contentious due to inherent differences in management characteristics and the controversy surrounding driving forces in rangeland vegetation dynamics. We used 500 m-long grazing gradients around water-points in order to evaluate the effects of grazing intensity on plant species composition and diversity, and to compare levels of degradation among management systems. We compared species composition, bare soil frequency, shrub, and tree density among management systems. We conclude that grazing has significant negative effects in these rangelands, although differences in degree of degradation could have been confounded by other events than grazing.

Additional keywords: rangeland degradation; grazing gradients; vegetation change; pastoralism; South Africa

Introduction

Rangeland management in South Africa

Due to low precipitation most of South Africa's land surface is not suitable for cultivation (Tainton, 1999). About 72% of the total land area of South Africa is suitable only as rangelands for livestock and game (Tainton, 1999; Anonymous, 2002). Three main types of rangeland management systems can be found in South Africa: commercial livestock ranching, communal livestock ranching, and game ranching. These management systems differ in: (1) management structure, (2) animal diversity, (3) management of grazing resource, and (4) products (see Table 1). Commercial livestock ranching is a well-developed industry in South Africa. The red meat industry, which is part of commercial livestock ranching, contributes approximately 12 % to the gross value of agricultural products produced in South Africa (Anonymous, 2002). Game ranching has been a fast growing sector in South Africa, and has grown by 6.75% per annum since 1993 (Kieser, 2001; Tomlinson *et al.*, 2002). Game ranches focus on ecotourism, trophy hunting, and venison production. Communal livestock ranches are also important users of South African rangelands. Many rural South Africans maintain livestock on communal lands to compensate for low or nonexistent income. Income of people having livestock in

Table 1: *Differences in management structure and product production, and animal diversity and grazing resource between management systems.*

	Communal livestock	Commercial cattle	Game
Management structure	Multiple managers	Single manager	Single manager
Animal diversity	Many different species	Single species	Many different species
Management of grazing resource	Continuous grazing, diverse vegetation	Rotational grazing, uniform vegetation	Continuous grazing, diverse vegetation
Products	High quantity, big diversity of products mostly for personal use	High quality, single product for domestic and international markets	High variety, strong healthy, big animals for trophies or eco-tourism

communal areas is nearly always generated outside these areas (Everson & Hatch, 1999). Communal livestock owners usually do not intend to sell their animals on markets; most products are for their own consumption and only surpluses are occasionally sold on markets (Tomlinson *et al.*, 2002). Animals are used for a wide array of purposes, e.g. milk production, social exchange (lobola), slaughter, and draft purposes (Tapson & Rose, 1984).

Rangeland degradation

Most commercial livestock and game ranches are managed by rangeland managers with secondary education and even some degree of tertiary education. South African agricultural research institutes have a long history of rangeland management research and extension in commercial ranching areas. Communal livestock management has, however, largely been based on traditional management systems without the livestock owners having any formal training in animal husbandry or rangeland management. The lack of education has long been considered a major cause of the perceived mismanagement of communal rangelands (Ellis & Swift, 1988; Behnke & Scoones, 1992). However, lack of education is not likely to be the main cause of rangeland degradation on communal ranches. One of the complications communal livestock ranches face is that the ranches are often managed by more than one manager. The problem of multiple managers on a rangeland was discussed in the 'tragedy of the commons' (Hardin, 1968) and has often been quoted as the main reason for degradation on communal livestock ranches (Ellis & Swift, 1988; Ward *et al.*, 2000). The 'tragedy of the commons' reasons that it is more profitable for an individual to overstock the 'commons' (i.e. communal lands) because he derives the entire benefit from each additional animal but the cost is shared by all. Due to this phenomenon, a growing number of livestock will populate the rangeland and will eventually exceed its ecological carrying capacity, which will lead to rangeland degradation (Hardin, 1968). The theory has, however, been criticised; some argue that communal lands are not completely without management rules, as Hardin (1968) implies in his theory. Livestock owners may, to a certain extent, regulate their herd size but cannot always link this to a particular grazing strategy. This is because they are bound to written and unwritten rules that govern the use of the communal rangeland (Tapson, 1993; Everson & Hatch, 1999). For example, we know from herd counts in KwaZulu-Natal, South Africa, that communal herds have remained stable for years (Tapson, 1993). These figures do not indicate an uncontrolled growth of the livestock population as the tragedy

of the commons predicts, and nor do the figures point to a crash in the livestock population due to land degradation.

Traditionally, rangelands have been seen as equilibrium systems driven by biotic effects, as described by Clements' (1916) plant succession theory. These systems develop towards an equilibrium along a series of successive stages starting at the pioneer stage and eventually reaching a climax stage determined by the constraints of the environment. Following this notion, biotic events drive the development of the vegetation (Clements, 1916). Clements' (1916) plant succession theory formed the intellectual underpinning of the classic rangeland condition concept described by Dyksterhuis (1949). The rangeland condition concept is based on the notion that alleviation of grazing pressure will cause vegetation to develop, through successional forces, towards the climax stage (Dyksterhuis, 1949). Closely tied to this concept is the notion that there is an ecological carrying capacity for livestock that is determined by the availability and quality of vegetation at equilibrium (= climax). In a ground-breaking study, Ellis and Swift (1988) challenged the traditional notion of equilibrium vegetation dynamics, and the allied concepts of carrying capacity and climax vegetation structure, that has been used to manage African pastoral ecosystems. Based on their long-term study in the semi-arid rangelands of the Turkana district in Kenya, they argued that vegetation in these rangelands was not regulated by livestock density but rather by abiotic events such as drought.

Noy-Meir (1973) had previously noted that arid and semi-arid areas are characterised by high spatio-temporal variability in precipitation with low predictability. The systems described by Ellis & Swift (1988) and Noy-Meir (1973) are called non-equilibrium systems. They do not tend towards equilibrium due to the stochastic nature of the climate (Behnke & Scoones, 1992). Non-equilibrium systems lack the density-dependence between vegetation and herbivore populations so important to equilibrium systems (Ellis & Swift, 1988; Behnke & Scoones, 1992). Because environmental variation causes large inter-annual variation in vegetation availability in such systems, animal numbers cannot easily track plant biomass. Thus, density-dependent effects of grazers on the vegetation are rare or non-existent. Therefore, following this view, herbivores cannot cause serious long-term rangeland degradation in such systems (Ellis & Swift, 1988; Behnke & Scoones, 1992). Non-equilibrium systems are thought to develop in a fundamentally different manner to equilibrium systems in that they do not progress in a linear fashion from pioneer to climax stages. One of the most prominent of the approaches used to describe vegetation change in non-equilibrium systems is the state-and-transition

model (Westoby *et al.*, 1989a, 1989b). Here, vegetation development is caused by an event that triggers a transition towards another vegetation state that may persist for a long time, or even be irreversible (Westoby *et al.*, 1989a, 1989b; Milton & Hoffman, 1994). The new approaches to rangeland dynamics have motivated many studies examining the effect of livestock densities on rangeland vegetation (see e.g. Abel, 1993; Scoones, 1993; Ward *et al.*, 1998; Dougill *et al.*, 1999; Fernandez-Gimenez & Allen-Diaz, 1999; Illius & O'Connor, 1999; Archer & Stokes, 2000; Cowling, 2000; Fynn & O'Connor, 2000; Van de Koppel & Rietkerk, 2000; Ward *et al.*, 2000). These studies also brought about a critical re-evaluation of the applicability of the carrying capacity in arid and semi-arid rangelands. Livestock numbers on communal ranches often exceed the calculated carrying capacity. Consequently, programs were initiated to reduce animal numbers in communal areas (Abel, 1993; Scoones, 1993). However, criticism of the carrying capacity concept and its applicability to non-equilibrium systems has been increasing. Stocking rate is mostly based on the primary productivity of a rangeland, which is closely related to rainfall in arid and semi-arid rangelands (Tapson, 1993). Because rainfall is so variable in arid and semi-arid rangelands (Noy-Meir, 1973), appropriate stocking rates would be so variable that they would become impracticable to implement (Bartels *et al.*, 1993; Tapson, 1993). Furthermore, there are many types of carrying capacities, of which the ecological and economical carrying capacities are the most important. Recommended carrying capacities are often based on economic carrying capacities (Jones & Sandland, 1974; Scoones, 1993), which is the stocking density that optimises economic return. The economic model recognises that economic return increases as livestock density increases and then declines after a certain optimum due to declining reproductive output and growth rates caused by competition among animals (Jones & Sandland, 1974). Exceeding economic carrying capacities does not imply rangeland degradation because the stocking density at which the animals start to compete (and at which productivity declines) is not consistent with the density at which degradation of the vegetation occurs (largely because vegetation change occurs at a slower pace than animal productivity change). Ecological carrying capacities, on the other hand, are determined on the basis of the ability of the vegetation to sustain the grazing intensity and should, if exceeded, result in degradation. Carrying capacities are, therefore, dependent on management objectives (Behnke & Scoones, 1992; Bartels *et al.*, 1993). Following the non-equilibrium view of rangelands, neither economic nor ecological carrying capacities is relevant to the topic of degradation because degradation in arid and semi-arid rangelands caused

by high stocking rates is 'reversed' by rainfall events (Ellis & Swift, 1988; Tapson, 1993; Ward *et al.*, 1998) because of the weak density-dependent coupling of animals to vegetation and the absence of some plant species from the sward in certain years (hence, grazing cannot affect them in these years) in arid and semi-arid systems.

Illius & O'Connor (1999) have criticised the non-equilibrium view and consider there to be no dichotomy between equilibrium and non-equilibrium systems. Illius & O'Connor (1999) concur with Ellis & Swift (1988) and Behnke & Scoones (1992) that rainfall is an important driving factor in arid and semi-arid rangelands. They also believe that there are areas in these systems that have weak or no density-dependent coupling of herbivores and vegetation. However, they do not agree that there is no density-dependent coupling. Illius & O'Connor (1999) argue that herbivore populations are coupled to key resources. Vegetation in arid and semi-arid areas is spatially heterogeneous in both amount and quality. Such heterogeneity is often associated with hydrological features, such as drainage lines, pans, and evergreen trees (Illius & O'Connor, 1999). Herbivore populations depend on these key resources for survival during times of drought and forage shortage. Herbivore populations are limited by the availability of key resources in a density-dependent manner (Illius & O'Connor, 1999). Because herbivores mainly depend on key resources, herbivore impacts are intensified in and directly around these key resources, which results in degradation of these areas caused by herbivore activity. Illius & O'Connor (1999) cite several studies where herbivores negatively affected key resources. Among other effects, studies found that vegetation of key resources changed from being dominated by perennial grasses to vegetation dominated by annual grasses. Illius & O'Connor (1999) believe that supporters of the non-equilibrium theory place too much emphasis on the effects of rapid vegetation change in response to abiotic events such as rainfall. Abiotic events most certainly have an affect on vegetation but in the long-term there is no net change, e.g. a good rainy period may compensate for the negative effects of a long drought (Illius & O'Connor, 1999). This is different for biotic factors, such as grazing as these factors nearly always have the same effect on a particular species (because of food preferences, for example). Although annual changes are small, when sustained over a long period they accumulate and may have long-term effects on vegetation composition (Illius & O'Connor, 1999).

Grazing gradients

We used grazing gradients to compare the effects of commercial cattle, communal livestock, and game ranching on rangeland vegetation. It is important to test the many generalisations about the different management systems. Generalisations such as the ones mentioned above have been proven to be wrong in some cases. Each case should therefore be empirically tested before one can label a certain management system as degrading to the particular rangeland (Behnke & Abel, 1996). An example where assumptions about communal livestock ranching did not materialise was a study conducted by Ward *et al.* (1998). The study compared rangeland degradation on commercial and communal ranches in Namibia. On both management types, soil and vegetation variables were collected and analysed. The analysis showed no difference between the two management types (ranches). The study argues that no difference can be detected between the ranches because of "the high inherent variability in rainfall [that] tends to mask the relative smaller impacts by pastoralism" (Ward *et al.*, 1998:369), although they do not exclude the possibility of long-term degradation due to herbivore activity (Ward *et al.*, 1998).

Studying the effect of herbivore activity on arid and semi-arid rangeland ecosystems has been proven to be difficult (e.g. Ellis & Swift, 1988; Behnke & Scoones, 1992). A number of problems arise when comparing effects of grazing regimes in arid and semi-arid rangelands: The spatial heterogeneity of precipitation in arid and semi-arid rangelands makes comparisons between management systems extremely difficult. Ellis & Swift (1988) argued that heterogeneity in precipitation is a major cause of the low impact of herbivory on arid and semi-arid rangelands. Wiens (1984) believes that there is a continuum between ecosystems where herbivore activity is the main driver of ecosystem change, and on the other hand ecosystems where heterogeneity of precipitation is the main driver of ecosystem change. Determining the most important cause of ecosystem change is difficult and adds to the complicated task of comparing grazing regimes in arid and semi-arid rangelands.

Fenceline contrasts (i.e. comparisons of the effects of different stocking levels or management types across a shared fenceline) have been used in the past (e.g. Ash *et al.,* 1995) to overcome some of the problems in assessing the impacts of herbivory mentioned above. However, fenceline contrasts are problematic when comparing continuous and rotational grazing systems (in our case, commercial cattle are run under rotational grazing and communal grazing is continuous) because differences may

be due to recency of use or timing of use only. Grazing gradients have proved to be an effective way of examining the effects of different management systems because similar gradients are compared (Lange, 1969; Tolsma *et al.*, 1987; Andrew, 1988; etc.). Near to the water-points, grazing intensity is highest and declines with distance. We can, therefore, correlate degradation indices directly to grazing intensity. Grazing gradients, also known as piospheres, were first used to test the effect of grazing on rangeland vegetation in Australia (Lange, 1969), and later in many other studies (e.g. Tolsma *et al.*, 1987; Andrew, 1988; Pickup & Chewings, 1994; Jeltsch *et al.*, 1997; Moleele & Perkins, 1998; Thrash, 1998; Dougill *et al.*, 1999; Saltz *et al.*, 1999; Thrash, 2000; Makhabu *et al.*, 2002; Riginos & Hoffman, 2003). We hypothesize that vegetation indicators such as species composition, shrub density and grass species abundance are able to show grazing impact around water-points.

To exclude short-term vegetation change caused by variable rainfall, we focus on indicators of long-term degradation in vegetation change. There are many indicators of long-term degradation due to herbivore activity; indicators vary from vegetation composition to soil quality. Among others, the following indicators have often been used in rangeland degradation studies: (1) Percentage perennial and annual grass species (Whitford *et al.*, 1998). Perennial grasses ensure a stable supply of forage over growing seasons, whereas the abundance of annuals is more variable due to climatic variability. (2) Percentage decreaser, increaser, and invader species. The classification of species into decreaser, increaser and invader species has its roots in the 'range condition' concept (Dyksterhuis, 1949). Here, decreasers are species that decrease in abundance due to heavy grazing; whereas, increaser species increase under the influence of heavy grazing; and invaders, invade and become dominant due to heavy grazing. Often the increaser group is divided in three subgroups (i.e. increaser I, II, III). Increaser I species are often abundant in under-utilised rangeland; increaser II species are abundant in overgrazed and disturbed rangelands with low precipitation; increaser III species are abundant in areas with higher rainfall where palatable species are suppressed by selective grazing (Van Oudtshoorn, 1999). The abundance of the groups gives an idea of the effect of grazing on the rangeland. (3) Bush density is often used as an indicator of bush encroachment (Tolsma *et al.*, 1987; Moleele & Perkins, 1998; Dougill *et al.*, 1999). Bush encroachment reduces the ability to provide forage for cattle because unpalatable bush replaces palatable grass. (4) Bare soil has a negative effect on ecosystem functioning. It reduces water infiltration, increases runoff, compaction, and loss of soil nutrients (Whitford *et al.*, 1998; De Soyza *et al.*, 2000).

Management systems may differ in grazing system and grazing pressure, but also in herbivore diversity. Various herbivore species may occupy different niches in a system. For example, ruminants are very efficient digesters compared to non-ruminants (Holechek *et al.*, 1995). Ruminants select high quality forage and become less efficient when forage has a high cellulose content, while non-ruminants are forced to be less selective due to their lower digestive efficiencies (they must ingest more plant biomass to retrieve the same nutrient intake as ruminants) and, consequently, are often obliged to subsist on lower quality forage (Holechek *et al.*, 1995). When a management system only integrates grazers in its system, only a part of the vegetation may be utilised (Ritchie & Olff, 1999). Single species systems (such as commercial cattle ranching) can have dramatic negative effects on vegetation composition due to selective grazing (Ritchie & Olff, 1999). A herd with many different species may actually have a positive compensatory effect by releasing plant species from competition while consuming different plant species, altering nutrient cycles, and disturbing soil (Ritchie & Olff, 1999).

Procedures

Study site

The study site is located 30 km north-west of Kimberley (Northern Cape, South Africa) and focuses on Pniel Estates near Barkly West. Coordinates of the approximate centre of the study site are 28°36' S, 24°28' E (1 125 m a.s.l.). The climate can be characterised as semi-arid with a mean annual precipitation of 388 mm (C.V. 39 %) (Kraaij, 2002) that mainly falls between November and April. Soils are mainly clayey along the banks of the Vaal River and near andesite outcrops. Further away from the river, soils are mainly deep red Kalahari sands with scattered calcrete pans (shallow, seasonally-inundated water bodies). The vegetation is classified as Kalahari thornveld (Acocks, 1988), or Kimberley thorn bushveld (Low & Rebelo, 1998) which includes *Acacia erioloba* and *Acacia tortilis* savanna, and *Acacia mellifera* shrubland. Dominant grasses are *Aristida congesta*, *Eragrostis lehmanniana*, and *Schmidtia pappophoroides*. The Estates are about 25 000 ha and are comprised of:

1. 3 000 ha communally-managed livestock ranch; 320 Large Stocking Unit (LSU) (9.375 ha/LSU).
2. 10 000 ha commercial game ranch; 912 LSU (10.965 ha/LSU).

3. 12 000 ha commercial cattle ranch; 750 LSU (16 ha/LSU).

Five properties border the Estates; viz. Vaalbos National Park, Rooipoort game ranch (owned by De Beers mining company), Platfontein communal game ranch (owned by the !Xu and Khwe communities), and two private owned cattle ranches (Secretaris and Nooitgedacht). The Estates are bounded on the northern side by the Vaal River.

We compared vegetation gradients around water-points in three different management systems, i.e. communal livestock ranching, commercial cattle ranching, and game ranching, to detect rangeland degradation. For each management system, we chose three water-points distributed over the particular ranch. To avoid possible confounding by soil type, all water-points selected were on clayey soils. At each water-point, three 500 m transects starting at the water-point were laid out. Vegetation parameters were sampled at fixed distances from the water-point (viz. 0, 25, 50, 75, 100, 150, 200 and every 50 m thereafter up to 500 m). In each management system, an additional water-point was sampled outside the Estates. For communal management, this was Camp 11 of the Barkly West

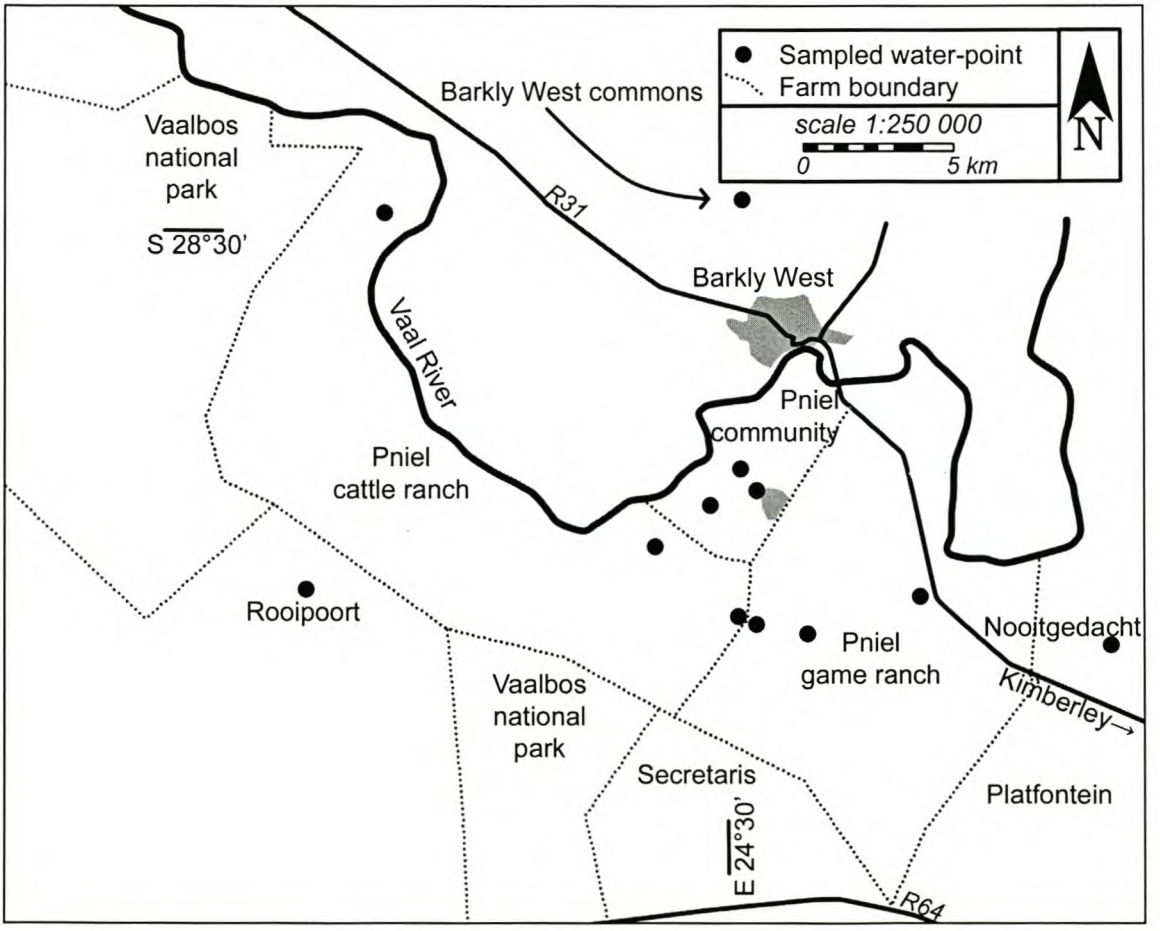

Figure 1: *Map of the study site.*

commons (coordinates: 28°29'27.56" S, 24°30'02.50" E), for commercial management we chose Nooitgedacht (coordinates: 28°38'21.08" S, 24°36'18.75" E), and for game management, Rooipoort game ranch (coordinates: 28°36'51.89" S, 24°21'54.54" E) was used. In total, nine water-points were sampled on Pniel Estates and three on other ranches (Figure 1). Next to the parameters discussed in this article, soil parameters were also collected along the same transects at the same time. The findings from these parameters are discussed in Smet & Ward (2003).

Vegetation composition and height

Vegetation composition and height were measured using a point-frequency frame of 1 m high and 1 m long. The frame has 10 holes (every 10 cm) through which a steel rod is lowered. Where the rod first touches a plant, the species and height are recorded (Mueller-Dombois & Ellenberg, 1974). This procedure was repeated three times at each of the fixed distances along the transect (once on the transect and once 5 m left and 5 m right of the transect). Using this method, 30 points were sampled at each distance along the transect. The point-frequency method gives an estimation of vegetation composition and vegetation height (Mueller-Dombois & Ellenberg, 1974).

Shrub and tree densities

We used the point-centred quarter method to determine the shrub and tree densities along the 500 m-long transects at the previously-mentioned distances. The distance from the sample point (at a particular distance from the water-point) to the centre of the nearest shrub and tree was measured and the species was determined in each of four quadrants (Mueller-Dombois & Ellenberg, 1974). This method gives an estimation of (1) density of tree or shrub species, (2) the dominance of tree or shrub species, and (3) the frequency of tree and shrub species (Mueller-Dombois & Ellenberg, 1974).

Statistical analysis

A number of variables were analysed for change along the transects and comparison among management systems. These variables were: species diversity and composition, vegetation height, shrub and tree composition and density, and the occurrence of bare patches. In addition, the abundance of three key grass species, *Schmidtia pappophoroides*, *Eragrostis lehmanniana*, *Aristida congesta* was analysed. These three grass species were common along all the transects. *Schmidtia pappophoroides* is a perennial grass, with a high grazing value; it is classified as a climax grass, and a decreaser species (Van Oudtshoorn, 1999). *Eragrostis lehmanniana* is a perennial grass species with

an average grazing value; it is classified as a sub climax grass, and an increaser II species (Van Oudtshoorn, 1999). *Aristida congesta* is an annual grass species and has a low grazing value; it is classified as a pioneer grass, and an increaser II species (Van Oudtshoorn, 1999).

Canonical correspondence analysis (CCA) was used to analyse vegetation composition in relation to distance from the water-point and management type. CCA is a constrained ordination technique that uses environmental variables to order sites based on vegetation composition (Ter Braak, 1995). Environmental variables used in these analyses were distance from the water-point and management type.

Because some variables where not normally distributed, we tested the significance of the patterns using randomisation tests (Crowley, 1992). We used the statistical computer program *Resampling Stats* (Simon, 1995) to calculate a test statistic functionally similar to the F statistic in ANOVA (in this case, the sum of deviations from the grand mean) using resampling with replacement (Simon, 1995). This statistic was compared to the statistic calculated for the data. We performed 1000 resamples to calculate the *P*-value (= probability that a random calculation > than the actual calculation - Munzbergova & Ward, 2002).

Other normally-distributed variables were also tested using analysis of covariance (ANCOVA). In these tests, management type and distance from the water-point were categorical and continuous predictors, respectively. A significant value in ANCOVA indicates differences among management systems when distance from water is controlled for.

Results

Grazing gradients

Vegetation height, bare soil, key species abundance, shrub and tree density were all correlated with distance from water-point (Table 2). Figure 2a and 2b show the change along the transect in mean bare soil frequency (frequency is the number of bare soil counts per 30 observations) and mean vegetation height, respectively. Bare soil frequency was, as may be expected, high around the water-point after which it steadily decreased. After 75 m from the water-point, bare soil frequency stabilised and no significant change was detected.

Bare soil frequency along transects was also tested for differences between management systems. ANCOVA results indicated a significant higher frequency of bare soil along transects in the communal

livestock ranch than in both commercial cattle and game ranch (Table 2). No significant difference was found between game and commercial cattle ranches. Vegetation height had a similar trend to bare soil. Up to 75 m from the water-point a significant increase in vegetation height was found. Thereafter, height reached an asymptote. Other variables that were correlated to distance from the water-point were *Schmidtia pappophoroides* abundance and vegetation composition (CCA axis 2) (Table 2 and 3). *Schmidtia pappophoroides* abundance was low up to 200 m from the water-point, after which the abundance stabilized (Figure 3). Variables that did not show significant trends along the grazing gradients were *Aristida congesta* abundance, *Eragrostis lehmanniana* abundance, vegetation composition (CCA axis 1), species diversity, evenness, richness, and tree density (Table 2 and 3).

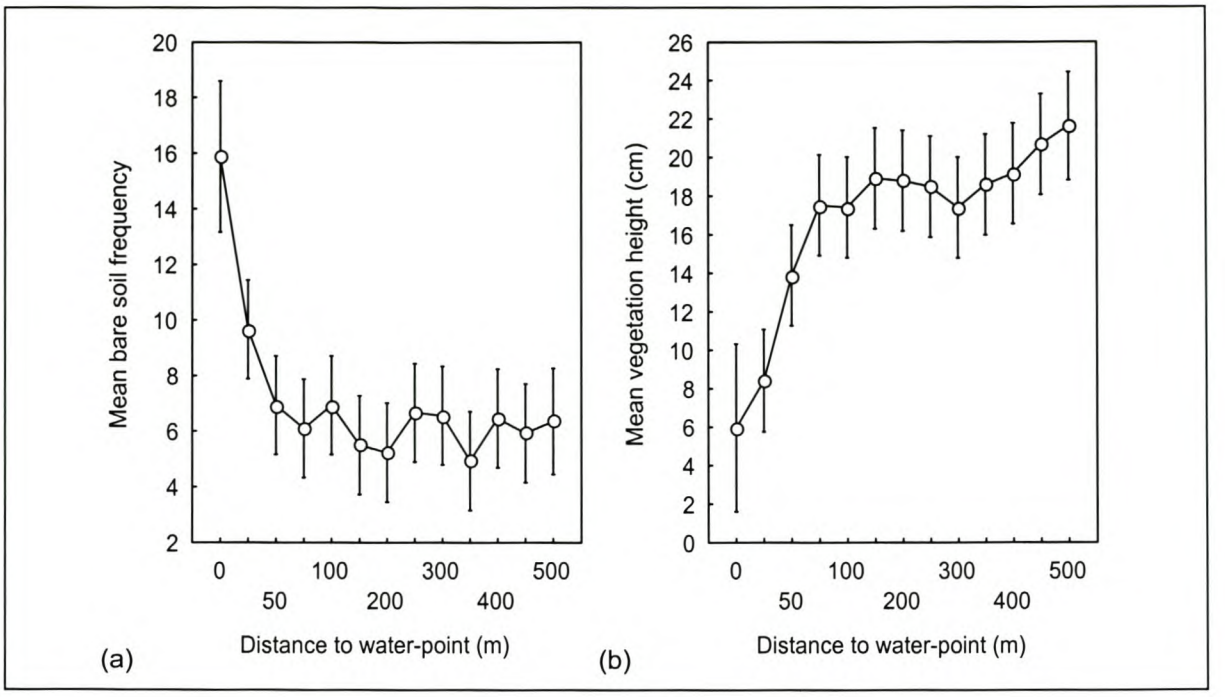

Figure 2: *(a) mean bare soil frequency (frequency is the number of bare soil counts per 30 observations) along the transects, (b) mean vegetation height along transects.*

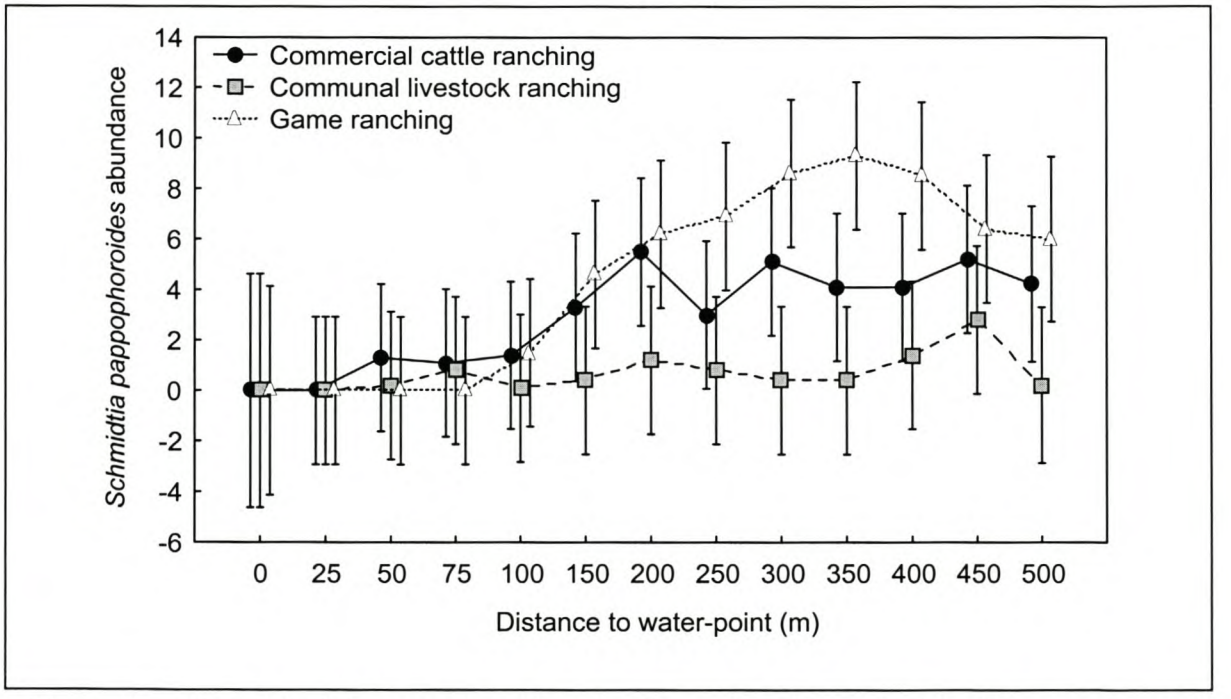

Figure 3: *Mean* Schmidtia pappophoroides *abundance along transects of the three management types (abundance is the number of* Schmidtia pappophoroides *counts per 30 observations).*

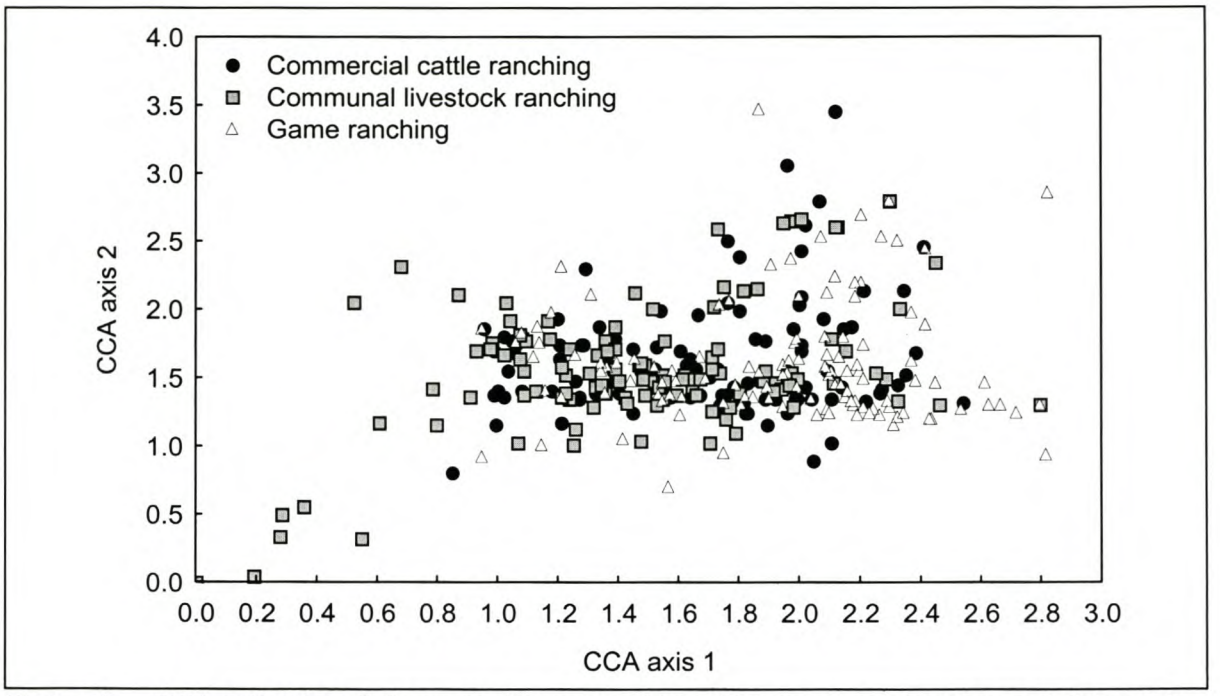

Figure 4: *CCA ordination of site vegetation composition with management system and distance to water-point as environmental variables (eigenvalue axis 1: 0.053, axis 2: 0.005). Management systems were distributed over axis 1 as follows (from left to right): communal livestock ranching, commercial cattle ranching, and game ranching.*

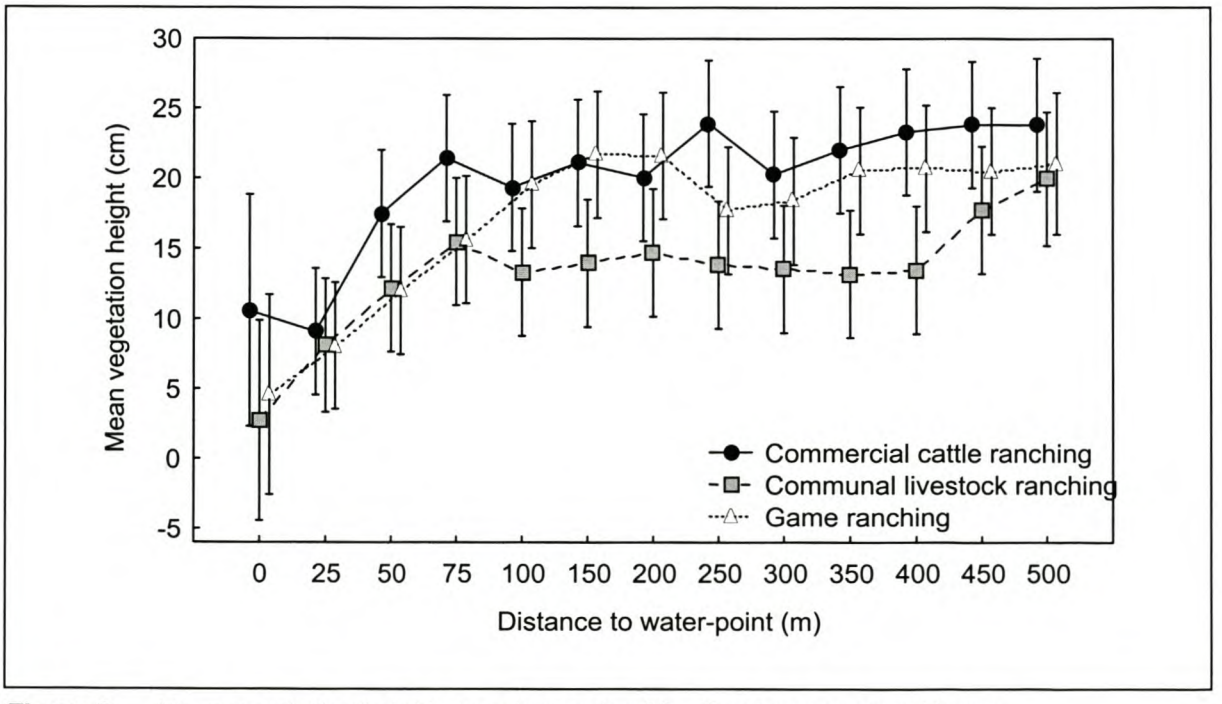

Figure 5: *Mean vegetation height along transects of the three management types.*

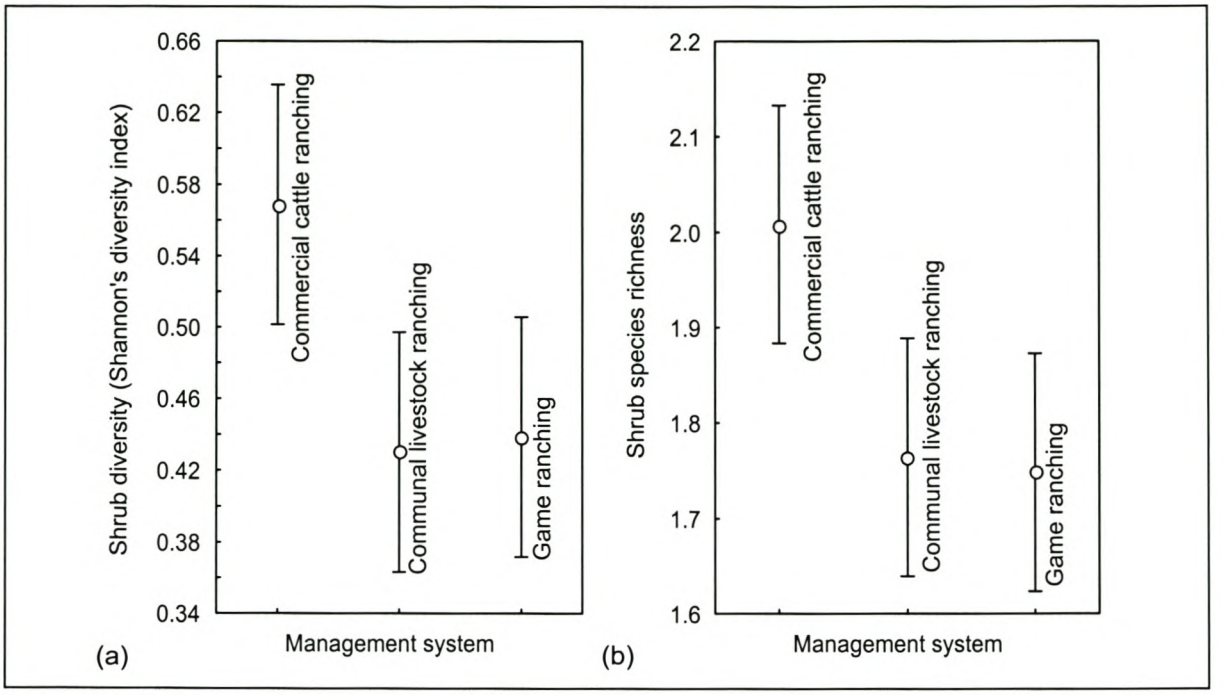

Figure 6: *(a) Shannon's diversity index for shrub diversity; (b) Shrub species richness.*

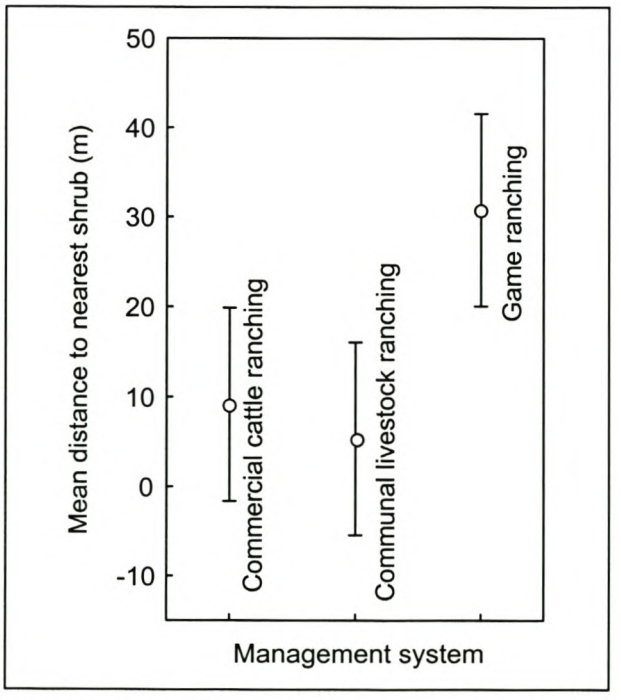

Figure 7: *Mean distance to nearest shrub (shrub density).*

Table 2: *ANCOVA results of various dependent variables. Bare soil frequency is the number of bare soil counts per 30 observations); diversity is Shannon's diversity index (Krebs, 1989); richness is the number of different species recorded per 30 observations; density is the distance to the nearest shrub or tree in meters; key species abundance (number of counts per 30 observations).*

	ANCOVA				
	Categorical predictor (management type)		Continuous predictor / covariate (distance to water-point)		
Dependent variable	F	p	F	p	df error
Bare soil	15.822	< 0.001	12.736	< 0.001	365
Vegetation height	22.712	< 0.001	51.317	< 0.001	362
Diversity	0.090	0.914	1.494	0.222	364
Richness	0.812	0.445	1.542	0.215	354
Key species abundance					
A. congesta	23.542	< 0.001	1.414	0.235	365
E. lehmanniana	7.468	< 0.001	1.137	0.287	365
S. pappophoroides	21.322	< 0.001	40.903	< 0.001	365
Shrub					
Diversity	5.174	0.006	0.099	0.754	365
Richness	5.281	0.005	0.126	< 0.001	365
Density	6.316	0.002	15.660	< 0.001	365
Tree					
Diversity	2.554	0.079	2.127	0.145	365
Richness	0.784	0.457	0.005	0.943	365
Density	23.012	< 0.001	11.068	< 0.001	365

Vegetation composition was analysed by CCA using management system and distance to water-point as environmental variables. Results from the CCA did not show high eigenvalues (axis 1: 0.053, axis 2: 0.005), which means that little of the variance in vegetation composition is explained by either of the axis (axis 1: 1.104 %, axis 2: 0.096 %) (Figure 4). However, there were significant differences among management systems on axis 1 (randomisation test: $p < 0.01$, df error = 999), but no significant effect of distance to water-point (Table 3). There was no significant difference among values on axis 2 between management systems ($p = 0.445$, df error = 999) but there was a significant effect of distance to water point (Table 3). Thus, it appears that axis 1 (the most important axis – Gauch, 1982) differentiates sites according to management strategy and axis 2 differentiates sites according to distance from water.

When examining individual species, trends were found for three grass species. Using ANCOVA, we found significant differences among management systems in *Aristida congesta*, *Eragrostis lehmanniana*, and *Schmidtia pappophoroides* abundance (Table 2). *Aristida congesta* was significantly more abundant along transects of the communal livestock ranch than in the other two management systems. The game ranch had a non-significantly lower abundance of *Aristida congesta* than the commercial cattle ranch. Similar effects were found among management systems and *Eragrostis lehmanniana* abundance. ANCOVA showed that commercial cattle ranching, followed by game ranching, had a significantly higher abundance of the grass species than communal livestock ranching. *Schmidtia pappophoroides*, however, was more abundant along the game transects, followed by commercial cattle ranching and then communal livestock ranching, which had the lowest abundance of the grass (Figure 3).

Vegetation height was also analysed using ANCOVA. Vegetation height was highest in the

Table 3: *ANCOVA results of CCA (canonical correspondence analysis), values do not have units.*

	ANCOVA				
	Categorical predictor (management type)		Continuous predictor / covariate (distance to water-point)		
Dependent variable	F	P	F	p	df error
CCA axis 1	27.318	< 0.001	1.638	0.2014	361
CCA axis 2	0.622	0.538	9.350	0.002	361

commercial cattle ranch, followed by the game ranch and lowest along transects in the communal livestock ranch (Figure 5 and Table 2). Vegetation height stabilised under Communal livestock ranching of between 75 and 400 m, after which it increased again. This was, however, different for the other management systems, which once vegetation height stabilised, it did not show further increases.

Tree and shrub composition and density

Tree species composition did not differ among management systems and was not affected by distance to water-point. Shrub composition did vary, however. Commercial cattle ranching had more diverse and richer species composition than communal livestock or game ranching (Figure 6a and 6b).

Tree and shrub density did not significantly differ among management systems, or along transects (Table 2). Overall tree and shrub density did show significant effects. This was highest in the communal livestock ranch whereas the game ranch had the lowest density of trees. Shrub densities in communal livestock and commercial cattle were not different. The game ranch had a significant lower density of shrubs (Figure 7) than other management systems.

Discussion

Grazing gradients

Several vegetation variables recorded in this study showed clear grazing gradients around all water-points. Additionally, distinct gradients in the amount of bare soil, which is known to be a good indicator of (poor) rangeland health (De Soyza *et al.*, 2000), were found in the piospheres. The gradients indicated the existence of two characteristic zones: (1) 0-75 m from the water-point, and (2) 100-500 m from the water-point. The existence of different zones has been recorded in previous studies: Jeltsch *et al.* (1997) found a sacrifice zone, also called an extreme degradation zone (James *et al.*, 1999), within 150 m from water-points in the Kalahari. Tolsma *et al.* (1987) found a similar zone within 200 m around water-points in Botswana. The denser shrub zone found in this study, after approximately 100 m in the second zone, is also comparable to other studies (Tolsma *et al.*, 1987; Andrew, 1988; Jeltsch *et al.*, 1997; James *et al.*, 1999). Vegetation height also showed these zones. However, vegetation height is probably merely an indication of grazing pressure and not degradation. Soil properties and nutrients showed similar zones as described above, indicating disturbance near to the water-points and decreasing disturbance further away from the water-points (Smet & Ward, 2003).

Our study was not able to identify a third zone that has been found to start beyond 1 km from the water-point in other studies (Jeltsch *et al.*, 1997). Compared to studies conducted in Australia (Bastin *et al.*, 1993; Pickup & Chewings, 1994), the grazing gradients we found were much shorter. This might be explained by differences in camp size (= fenced grazing area/rangeland) and herd size which are generally bigger in Australia than in South Africa. Maximum distance to a water-point was approximately 1 km in an average camp, whereas in Australia, Pickup & Chewings (1994) used grazing gradients of 10 km from water-points. Thus, a possible reason for our inability to identify a third zone may be that 500 m transects were not long enough to identify a third zone (due to the limitation in camp size).

Management systems and grazing gradients

We found significant differences in bare soil, shrub density, *Aristida congesta*, *Eragrostis lehmanniana*, and *Schmidtia pappophoroides* abundance among management types. These are all indicators of long-term degradation (see above). We were especially interested in the difference in degradation between the management types. We found high abundance of *Aristida congesta* and low abundance of *Eragrostis lehmanniana* and *Schmidtia pappophoroides* around the water-points of the communal livestock ranch compared to those around commercial cattle and game ranches. *Aristida congesta* is an annual grass species known to have low palatability (Van Oudtshoorn, 1999). Moreover, the grass species is generally found in degraded areas (Van Oudtshoorn, 1999). Shrub densities along transects in the communal livestock ranch were higher than along the transects of the other two management systems. This may be caused by the high stocking density in the communal livestock ranch (9.375 ha/LSU). The increase of shrubs density is called bush encroachment (Smit *et al.*, 1999). The cause of bush encroachment has been a contentious issue in rangeland ecology. Many events have been suggested as main cause of the phenomenon: heavy grazing (Andrew, 1988), fire (Westoby *et al.*, 1989), soil nutrient change (Noy-Meir, 1982), and rainfall (Kraaij, 2002; Ward, 2003). However, most authors agree that bush encroachment has caused a decline in grazing capacity of large areas in the South African savannas (Smit *et al.*, 1999).

Unlike the commercial cattle ranches, the communal livestock ranches have different herbivores: grazers (cattle) and browsers (goats), ruminants (e.g. cattle, sheep) and non-ruminants (e.g. horses, donkeys). These herbivores have different digestive systems. Feeding strategies are adapted to the digestive system of the particular herbivore (Owen-Smith, 1999). Having different herbivores

increases the ability of a management system to exploit the different vegetation types in a rangeland (Owen-Smith, 1999). Vegetation change might have negative effects on the production of one herbivore species, yet it might be advantageous for another. In effect, herbivore diversity might have a stabilising effect on the productivity of a rangeland with a continuously changing vegetation, as might be experienced in non-equilibrium systems.

The commercial cattle ranches have only one herbivore species (cattle). As discussed above, this means that any vegetation change may potentially affect the productivity of these ranches. However, due to the relatively low stocking densities (16 ha/LSU) and the rotational grazing system applied in these ranches, serious negative affects have not yet been recorded. Shrub densities were relatively high around the water-points, but the perennial grass species *Eragrostis lehmanniana*, and *Schmidtia pappophoroides*, are still dominant grasses. These two species are preferred forage species for cattle (Van Oudtshoorn, 1999), and are dominant in natural Kalahari thornveld (Acocks, 1988). No clear sign of a change in grass species composition was found along any of the transects in this management type. However, due to the fact that the commercial cattle ranches only accommodate cattle, selective grazing might cause long-term degradation which is not yet visible (Illius & O'Connor, 1999).

The high stocking rate (10.965 ha/LSU) on the game ranches would suggest that vegetation under game ranching would be more affected than on the commercial cattle ranches. However, game ranching showed the least signs of rangeland degradation. Abundance of the perennial grass species, *Eragrostis lehmanniana* and *Schmidtia pappophoroides*, was highest (among management systems) and, conversely, shrub density was lowest along the game ranch transects. The game ranches need to have a high diversity of game to satisfy the needs of their customers. In the case of the Pniel Estates game ranch, more than 17 game species can be found. The game species found in the ranch range from large giraffe and rhinoceros to small steenbok. These game species all have their own specific feeding strategy. As mentioned above, this not only results in a much more efficient use of a diverse rangeland, but it also minimises selective grazing. Vegetation not eaten by a certain herbivore is likely to be eaten by another.

Vegetation dynamics

Climatic events are undoubtedly important driving factors of vegetation change in the semi-arid rangelands of the study area (Noy-Meir, 1973; Ellis & Swift, 1988). It is, however, unlikely that

herbivory does not have any effect on vegetation dynamics. In systems where key-resources are sufficiently reliable to ensure stable herd sizes, density-dependent processes are likely to occur. In our study area, the herbivore populations all have access to different key-resources. The communal livestock ranch of Pniel Estates borders the Vaal River, and grasses and reeds along the banks of the river may allow a continuous supply of forage during dry periods. Cattle in the commercial cattle ranch receive supplementary nutrients (lick) at water-points, as do animals on the game ranch, enabling them to survive the periods when forage is limited. Both in the game and communal ranches, *Acacia erioloba* woodlands on deep sands appear to serve as key resources that are preferentially used until vegetation availability reaches a low point in the dry season. We note in this regard that an early resident of Pniel Estates, Solomon Tshekisho Plaatje, recorded that the *A. erioloba* woodlands were heavily utilized by communally-managed livestock more than 100 years ago, indicating their relative resilience to heavy grazing pressure (Willan, 1996). A more recent study (Britz, 2003) has shown that communally-managed *A. erioloba* woodlands still produce a high biomass of palatable grasses in spite of prolonged heavy grazing but do have higher abundances of poisonous and unpalatable species than similar woodlands under game ranching. The trends in the vegetation data show that herbivores do affect rangeland vegetation. These results are consistent with Illius & O'Connor's (1999) discussion that arid and semi-arid rangelands are not excluded from herbivore-induced degradation even though high variability of climatic events is a major modifier of vegetation in these systems. Key-resources, such as the ones mentioned above, sustain herbivore herds that may cause rangeland degradation.

Conclusion

Grazing gradients are good indicators of the effects of rangeland management systems in spatially-heterogeneous semi-arid rangeland ecosystems. The main degradation indicators were found to be the abundance of three key grass species, shrub density and bare soil (and overall species composition in CCA). The indicators showed that grazing gradients around water-points in the communal livestock ranches had higher degrees of degradation than commercial cattle and game ranching. Game ranching had a positive effect on recovery of the rangeland, as water-points previously used for commercial cattle and converted into game water-points in 1989 showed fewer signs of degradation than the water-points currently used for commercial cattle. Here, high herbivore diversity in the game ranch is most likely to have a positive influence on rangeland condition.

The CCA clearly showed that management type has a more important effect on vegetation composition than varying grazing pressure along the transect. This can be concluded because the first axis calculated by a CCA is, by definition, the axis that explains the largest portion of the variance (Gauch, 1982). This implies that less variance is explained by difference in grazing pressure along the transect compared to management system. Because there were clear differences and patterns in degradation among management types, we believe that biotic events, such as grazing regime, herbivore species, herbivore diversity and grazing pressure, do influence vegetation dynamics in the study area. This finding leads us to conclude that our study area is not a true non-equilibrium system as described by Ellis & Swift (1988) and Behnke & Scoones (1992). The ideas discussed by Illius & O'Connor (1999), however, are plausible explanations for our results. As we have discussed in the discussion, all herbivore herds have some kind of key-resource as described by Illius & O'Connor (1999). Key resources in our study are either Vaal River riparian vegetation (for the communal ranches) or *Acacia erioloba* woodlands (game and communal ranches). The commercial ranches do not appear to have a natural key resource and supplementary nutrients must be provided.

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Soil quality gradients around water-points under different management systems in a semi-arid savanna, South Africa

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Abstract

Over 70% of South Africa is too arid for crop farming and is used for either commercial livestock ranching, communal livestock ranching, or game ranching. The inherently different management characteristics of these ranching systems and their effects on vegetation dynamics makes rangeland degradation a contentious issue. We used 500 m grazing gradients around water-points to evaluate the effects of management type on soil quality. Results showed significant negative effects of management type on soil parameters (i.e. soil pH, nitrogen, and organic carbon) within 0 – 100 m from the water-point. Commercial livestock ranching had the greatest negative effect on the immediate area around the water-point. Beyond 100 m, no effect of herbivore activity on soil parameters was detected under any management system.

Keywords: semi-arid rangeland; pastoralism; wildlife; rangeland degradation; soil condition; grazing gradients

Introduction

Rangelands are important ecosystems for South Africa. More than 75% of South Africa's land surface is used for pastoral production (Tainton, 1999). This is mainly due to the low precipitation in these areas, which makes the land unsuitable for crop farming. Three main rangeland management systems can be found in South Africa: commercial livestock ranching, communal livestock ranching, and game ranching. These three systems differ in management structure, animal diversity, management of grazing resources, and products (Table 1).

Of the three livestock management systems mentioned above, communal livestock ranching has most often been criticised for its perceived degrading effects on rangelands (Ellis & Swift, 1988; Behnke & Scoones, 1992). In the 'tragedy of the commons' (Hardin, 1968), it is argued that rangeland degradation of communal lands is nearly inevitable due to the many individuals that utilise the same land. The theory reasons that it is beneficial for an individual livestock owner to be selfish and overstock the 'commons' (i.e. communal lands) rather than to apply conservative stocking rates because the benefits thereof accrue to the individual but the costs of heavy stocking are shared by all. This then leads to overstocking and eventually to irreversible degradation (Hardin, 1968). African communal rangelands frequently support high numbers of livestock, and often exceed advised carrying capacity levels (Abel, 1993; Tapson, 1993; Scoones, 1993; Ward *et al.*, 1998). Because of the high stocking rates on communal lands it has often been assumed that communal livestock

Table 1: *Differences in management structure, production, and animal diversity and grazing resource between management systems.*

	Communal livestock	Commercial cattle	Game
Management structure	Multiple managers	Single manager	Single manager
Animal diversity	Many different species	Single species	Many different species
Management of grazing resource	Continuous grazing, diverse vegetation	Rotational grazing, uniform vegetation	Continuous grazing, diverse vegetation
Produced products	High quantity, big diversity of products mostly for personal use	High quality, single product for domestic and international markets	High variety, strong healthy, big animals for trophies or eco-tourism

management systems cause rangeland degradation (Ellis & Swift, 1988; Behnke & Scoones, 1992; Ward *et al.*, 1998; Dougill *et al.*, 1999). Destocking programs have been initiated to prevent rangeland degradation but many have failed to reach their objectives (Abel, 1993). Contrastingly, commercial cattle ranchers apply much lower stocking rates in order to produce high quality products for markets.

Rangeland degradation

Traditionally, arid and semi-arid rangeland vegetation dynamics have been explained using equilibrium models (Ellis & Swift, 1988; Behnke & Scoones, 1992). Equilibrium models reason that vegetation dynamics are regulated by biotic controls such as herbivory and that vegetation tends towards an equilibrium (Clements, 1916). Vegetation moves in a predictable manner along a series of successional stages, beginning at the pioneer stage and ending at the climax stage. Change from one stage to another is driven by biotic events (Clements, 1916). It has long been assumed that grazing is an important modifier of rangeland vegetation, and therefore also a root cause of rangeland degradation where heavy stocking occurs (Dyksterhuis, 1949; Ellison, 1960; Stoddart, 1960; Holechek *et al.*, 1995). Management systems have long been based on this assumption, e.g. carrying capacity (Stoddart, 1960) and rangeland condition (Dyksterhuis, 1949) concepts have been important management tools to regulate grazing intensity and monitor grazing resources to prevent degradation (Joyce, 1993; Holechek *et al.*, 1995; Tainton *et al.*, 1999).

Ellis & Swift (1988) reviewed the issue of stability of African pastoral ecosystems, based largely on their long-term study in the Turkana region of Kenya. They questioned the widespread assumption that communal pastoralism has degrading effects on ecosystems. They argued it did not because of the highly variable nature of vegetation dynamics of arid and semi-arid rangelands occupied by most African communal pastoralists. Ellis & Swift (1988) argued that abiotic events had a much more important role to play in the vegetation dynamics of these systems than did grazing. It is well known that rainfall in arid and semi-arid ecosystems is highly variable in time and space (Noy-Meir, 1973). Ellis & Swift (1988) reasoned that these climatic attributes are the driving force behind vegetation dynamics; and not biotic events, such as grazing (Ellis & Swift, 1988; Behnke & Scoones, 1992). The stochastic nature of the climatic factors results in erratic vegetation changes and a situation where the vegetation is unlikely to reach an equilibrium state (Ellis & Swift, 1988; Westoby *et al.*, 1989; Behnke & Scoones, 1992; Milton & Hoffman, 1994). Therefore, these ecosystems are called non-equilibrium systems (Ellis & Swift, 1988; Behnke & Scoones, 1992). Changes in non-equilibrium vegetation have

been described by the state-and-transition models (Westoby *et al.*, 1989; Milton & Hoffman, 1994). These models allow certain events to cause transitions from one vegetation state to another, and once a transition has occurred, vegetation states may persist for a long period and may even be irreversible (Westoby *et al.*, 1989).

Non-equilibrium systems are thought to have weak or no density-dependent processes between vegetation and herbivore populations (Ellis & Swift, 1988; Illius & O'Connor, 1999). Traditional rangeland management tools, such as carrying capacity and the rangeland condition concept, are all based on density-dependence processes (Dyksterhuis, 1949; Jones & Sandland, 1974). These management tools would therefore be of little use in non-equilibrium rangelands. The non-equilibrium concept caused many rangeland scientists to re-evaluate the effect of stocking rates on arid and semi-arid rangelands (i.e. Behnke & Scoones, 1992; Abel, 1993; De Leeuw & Tothill, 1993; Illius & O'Connor, 1999). For example, Tapson (1993) found that herd size on KwaZulu's communal lands remained stable over the period 1974-1988 *contra* the conventional expectation that sustained heavy stocking had caused a decline in vegetation productivity, resulting in decreased animal production. Similarly, Ward *et al.* (1998) did not find a difference in degradation when comparing commercial ranches and a large (117 000 ha) communal ranch, differing 3-10-fold in stocking densities. This suggested that stocking densities did not influence rangeland degradation.

An exact border between equilibrium systems and non-equilibrium systems cannot easily be determined. Wiens (1984) suggests that rangelands are located somewhere on a continuum between equilibrium and non-equilibrium dynamics. Therefore, it is not always clear to what extent a system is dominated by either equilibrium or by non-equilibrium forces; and how successful traditional management tools are.

Illius & O'Connor (1999) discussed the relevance of non-equilibrium models for semi-arid rangelands. They concurred with Ellis & Swift (1988) and Behnke & Scoones (1992) that there is a need to review the applicability of equilibrium models to arid and semi-arid rangelands. However, they also believe that density-dependent processes are more important than Ellis & Swift (1988) and Behnke & Scoones (1992) portray. In their article, Illius & O'Connor (1999) argue that density-dependence can occur in arid and semi-arid rangelands. Arid and semi-arid rangeland vegetation is spatially heterogeneous. Vegetation associated with drainage lines, pans, and evergreen trees can function as key resources for herbivore populations (Illius & O'Connor, 1999). The availability of such

resources determines herbivore population size (Illius & O'Connor, 1999). There is a density-dependent relationship between herbivore population and key resources. During long droughts, this relationship becomes more apparent (Illius & O'Connor, 1999). They therefore believe that: "... spatially and temporally, the whole system is heterogeneous in the strengths of the forces tending to equilibrium, these diminishing with distance to watering and key-resource areas" (Illius & O'Connor, 1999:809). In other words, vegetation dynamics in key resource areas exhibit equilibrium dynamics, but outside these key resource areas exhibit non-equilibrium dynamics. Furthermore, they argue that "rainfall variability has profound effects on annual variation in species abundance, but, unless there is a directional trend in rainfall, and despite the potential for large annual changes, there is no net change in species composition in the long-term" (Illius & O'Connor, 1999:803). Grazing, on the other hand, may have only a small annual effect on species composition. However, the effects of grazing on a species do not change (due, for example, to forage preferences) and, therefore, grazing has a cumulative effect on species composition that can become substantial over long periods of intensive grazing (Illius & O'Connor, 1999).

Grazing gradients

The present study aims to compare rangeland degradation between three - management systems. Due to the uncertainty regarding the extent that semi-arid rangelands are regulated by equilibrium or non-equilibrium dynamics, rangeland degradation is a contentious issue. Grazing gradients can be used to correlate degradation to a particular management system (i.e. with a specific grazing intensity and herbivore species) in arid and semi-arid environments where vegetation and soil quality are spatially variable. Gradients starting at water-points show higher grazing intensity and rangeland degradation near to the water-points compared to further away from (Tolsma *et al.*, 1987). These grazing gradients are also called 'piospheres' (Lange, 1969). Many studies have used grazing gradients to evaluate the effect of grazing on vegetation (Walker & Heitschmidt, 1986; Tolsma *et al.*, 1987; Andrew, 1988; Jeltsch *et al.*, 1997; Moleele & Perkins, 1998; Thrash, 1998; James *et al.*, 1999; Thrash, 2000; Riginos & Hoffman, 2003). As discussed above, vegetation is profoundly affected by variations in rainfall (Ellis & Swift, 1988; Behnke & Scoones, 1992; Illius & O'Connor, 1999). Recording vegetation composition is, therefore, a snapshot of a short-term situation and does not necessarily reflect the long-term situation. In this study, we used soil indicators to determine rangeland health. Soil indicators are reliable indicators of long-term soil degradation because they are

not affected by short-term rainfall fluctuations and drought to the degree that vegetation composition is (Dougill *et al.*, 1999; Turner, 1999).

Methods and Materials

Study site

The study site is located 30 km north-west of Kimberley (Northern Cape, South Africa) and focuses on Pniel Estates near Barkly West, (Figure 1). The coordinates of the centre of the study site are: 28°36' S, 24°28' E (1 125 m a.s.l.). The climate can be characterised as semi-arid with a mean annual precipitation of 388 mm (C.V. 39 %) (Kraaij, 2002) that mainly falls between November and April. Soils are mainly clayey along the banks of the Vaal River and near andesite outcrops. Further away from the river, soils are mainly deep red Kalahari sands with some calcrete pans (shallow, seasonally-inundated water bodies). The vegetation is classified as Kalahari thornveld (Acocks, 1988) or Kimberley thorn bushveld (Low & Rebelo, 1998) which includes *Acacia erioloba* and *Acacia tortilis* savanna, and *Acacia mellifera* shrubland. Dominant grasses are *Aristida congesta*, *Eragrostis*

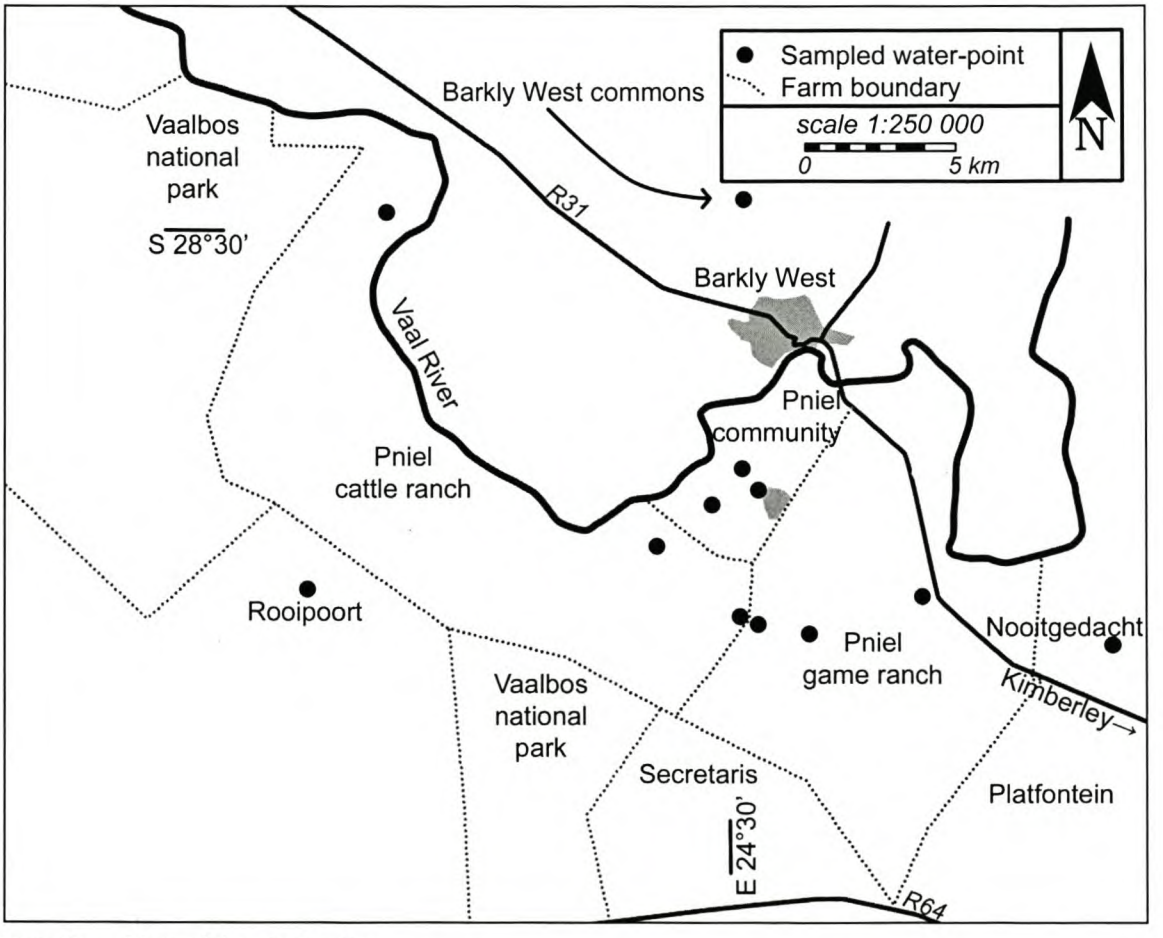

Figure 1: *Map of the study site.*

lehmanniana, and *Schmidtia pappophoroides*. Pniel Estates is about 25 000 ha in extent and is comprised of:

1. 3 000 ha communally-managed livestock ranch; 320 Large Stock Unit (LSU) (9.375 ha/LSU).
2. 10 000 ha commercial game ranch; 912 LSU (10.965 ha/LSU).
3. 12 000 ha commercial cattle ranch; 750 LSU (16 ha/LSU).

The Evangelical Lutheran Church of South Africa owns the estate, including the communal area. Five properties border the Estate; viz. Vaalbos National Park, Rooipoort game ranch, Platfontein communal game ranch (owned by the !Xu and Khwe San (Bushman) communities), and two privately-owned cattle ranches (Secretaris and Nooitgedacht). North of Pniel Estates lies the Vaal River (Figure 1).

For each management system, three water-points were chosen, distributed over the particular ranches and all on clayey soils. For each management system, an additional water-point was sampled outside Pniel Estates, for the communal ranch this was Camp 11 of the Barkly West commons (coordinates: 28°29'27.56" S, 24°30'02.50" E), for the commercial ranch we chose Nooitgedacht (coordinates: 28°38'21.08" S, 24°36'18.75" E), and for the game ranch the Rooipoort game ranch (coordinates: 28°36'51.89" S, 24°21'54.54" E) was used (Figure 1). In total, 9 water-points were sampled on Pniel Estates and 3 on other ranches. At each water-point, three 500 m transects were set out, radiating from the water-point. Soil samples were collected at fixed distances from the water-point (i.e. 0, 25, 50, 75, 100, 150, 200 and every 50 m thereafter to 500 m). At every distance, three soil samples were collected to a depth of 10 cm. The first sample was collected on the transect and the others 5 m left and right of the transect. The three samples were combined into one composite sample. Soils were dried (40 °C for 24 h) and kept dry at room temperature before being analysed. Next to the parameters discussed in this article, vegetation parameters were collected along the same transects at the same time. The results of the vegetation study are discussed in Smet & Ward (2003).

Soil analysis

Of the 12 water-points, only a sub-sample of soil samples collected along the transects was analysed. In each management system, two water-points on Pniel Estates and one outside Pniel Estates was chosen. At each of these water-points, one transect was selected at random, bringing the total number of transects to three per management system. The soil samples of these transects

were analysed by a commercial laboratory. The following soil nutrients and characteristics were analysed: soil pH_{KCl}, resistance, total nitrogen (N), available phosphorus (P), and organic carbon (C). Total soil N was determined using Dumas' total combustion method (Dumas, 1831; Bremner, 1996) with a Leco nitrogen analyser. Available P was analysed using the Bray II method (Bray & Kurtz, 1945). Organic C was determined using the Walkley-Black method (Walkley, 1947; Nelson & Sommers, 1996).

Soil bioassay

Each composite sample was transferred into a black plastic bag (dimensions: 19 cm high x 8 cm wide x 5 cm deep) after which the bags were tagged for identification. These were then randomly placed on a nursery bed. The grass species *Eragrostis tef* was sown in each of the bags. This species was chosen as it is closely related to species commonly found in our study (i.e. *Eragrostis lehmanniana*). *E. tef* is, therefore, likely to respond in similar ways as the natural occurring species do. Furthermore, the species is commercially available, and commercial seeds are more likely to be similar to each other than seeds collected from the rangeland. This reduces the risk that bio-assay results are biased because of variance in seed quality. Bags were watered twice daily using an automatic sprinkler system. Grasses were left to grow for two months before being harvested. Plants were washed and roots separated from the shoot. Roots and shoot were stored separately in paper bags and dried for 24 h at 70 °C. After drying roots and shoots, their respective dry masses were recorded.

Statistical analysis

Variables were tested using analysis of covariance (ANCOVA). In these tests, management type and distance from the water-point were categorical and continuous predictors, respectively. We tested various variables using broken-stick regression models. The procedure we used was as follows: We determined the approximate breakpoint of the regression by eye and then calculated the error M.S. for the linear regressions for the data points to the left and to the right of this point. We then calculated the pooled error M.S. for these two regression lines. After this, we shifted the breakpoint one sampling unit to the left of the initial breakpoint and recalculated the pooled error M.S. as above. Then we recalculated the pooled error M.S. using a breakpoint one sampling unit to the right of the initial breakpoint. The breakpoint with the lowest pooled error M.S. (i.e. the broken stick regressions explain

most of the variance in the data) was taken as the distance at which the water-point effect ended (see Ward & Pinshow (1994) for further explanation of this statistical analysis).

Results

Soil pH

There was a significant effect of management type on soil pH (ANCOVA: $F = 3.306$, $p = 0.040$, df error = 109). There was also a significant effect of the covariate (ANCOVA distance: $F = 44.017$, $p < 0.001$). Soils around the water-points in the communal livestock ranches had lower soil pH values than the other two management systems. However, there was no significant difference in the effects of commercial cattle and game ranching on soil pH (ANCOVA: $F = 0.051$, $p = 0.822$, df error = 71).

As can be seen from Figure 2, soil pH values were high around water-points, followed by a steady decrease in pH up to 100 m from the water-point. After this distance, soil pH reached an asymptote. The broken-stick regression confirmed the breakpoint to be at 100 m from the water-point.

Soil resistance

Management type did not have a significant effect on soil resistance (ANCOVA: F 2.962, p 0.056, df error 109). However, there was a significant effect of the covariate on soil resistance (ANCOVA distance: $F = 15.472$, $p < 0.001$). Soil resistance was low near water-points and increased to 100 m from the water-point (Figure 3). No further change was found beyond 100 m from the water-point.

Soil nitrogen

Management type had a significant effect on soil N (ANCOVA: $F = 4.932$, $p = 0.015$, df error = 109), as did the covariate (ANCOVA distance: $F = 13.874$, $p = <0.001$). Here, commercial cattle ranching had the highest soil N concentrations, whereas communal livestock and game ranching had comparable levels. Soil N concentration in the vicinity from the water-points (i.e. 0 - 75 m) in the commercial cattle ranches was many times higher than in the other two systems (Figure 4). At 75 m from the water-point, this returned to the same levels as under the other two management types.

Soil phosphorus

Soil phosphorus was not affected by management type (ANCOVA: $F = 2.169$, $p = 0.119$, df error = 109), but was affected by the covariate, distance (ANCOVA distance: $F = 16.773$, $p < 0.001$). Within

75 m of the water point there is a change in soil P concentrations, except for communal livestock farming where the change is minimal (Figure 5).

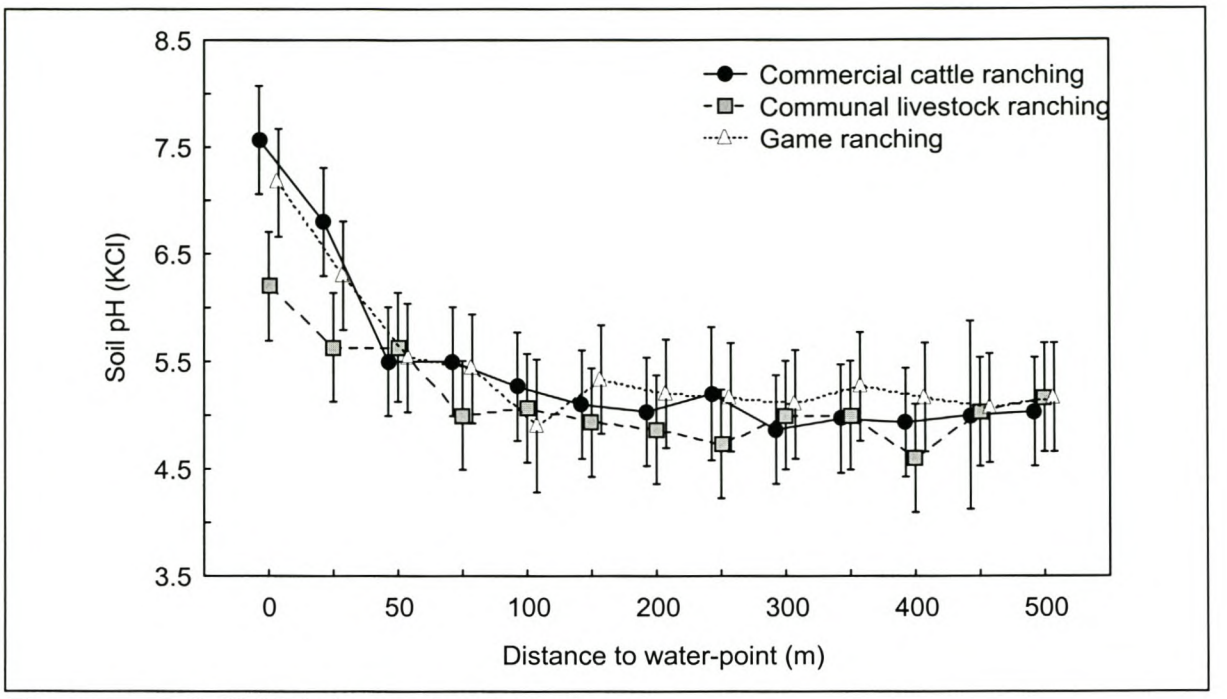

Figure 2: *Soil pH (KCl) change along transects.*

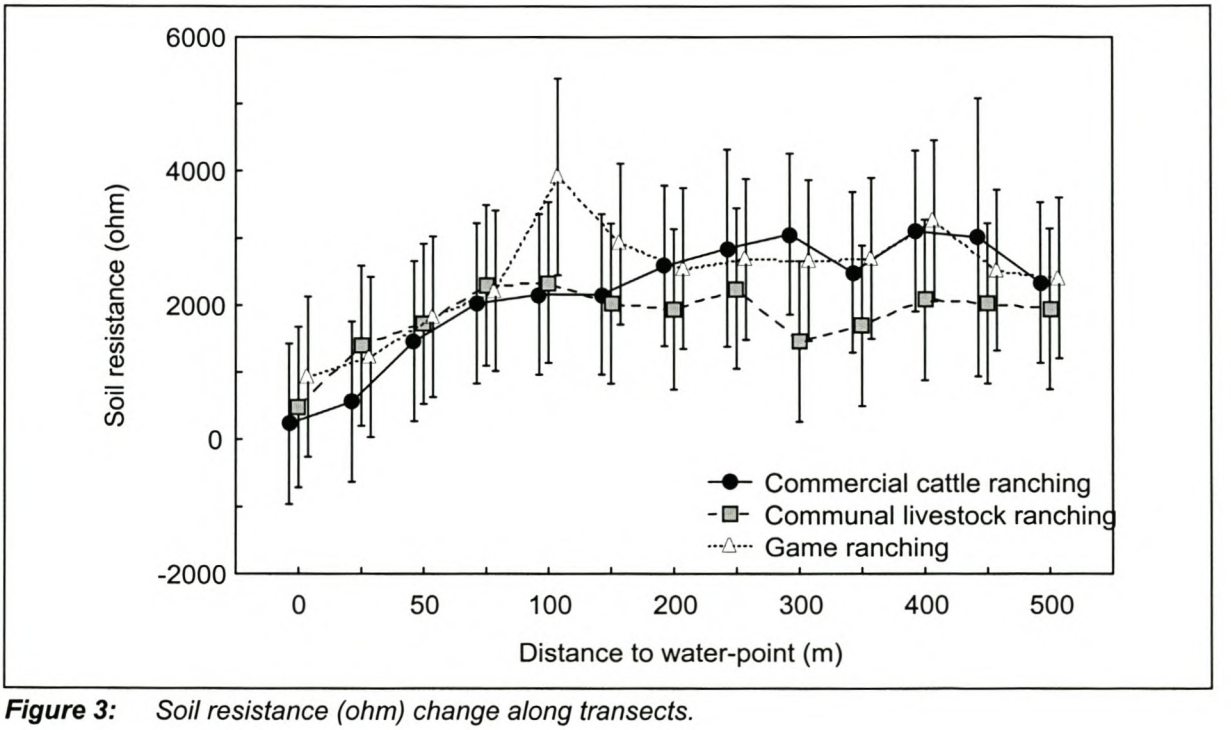

Figure 3: *Soil resistance (ohm) change along transects.*

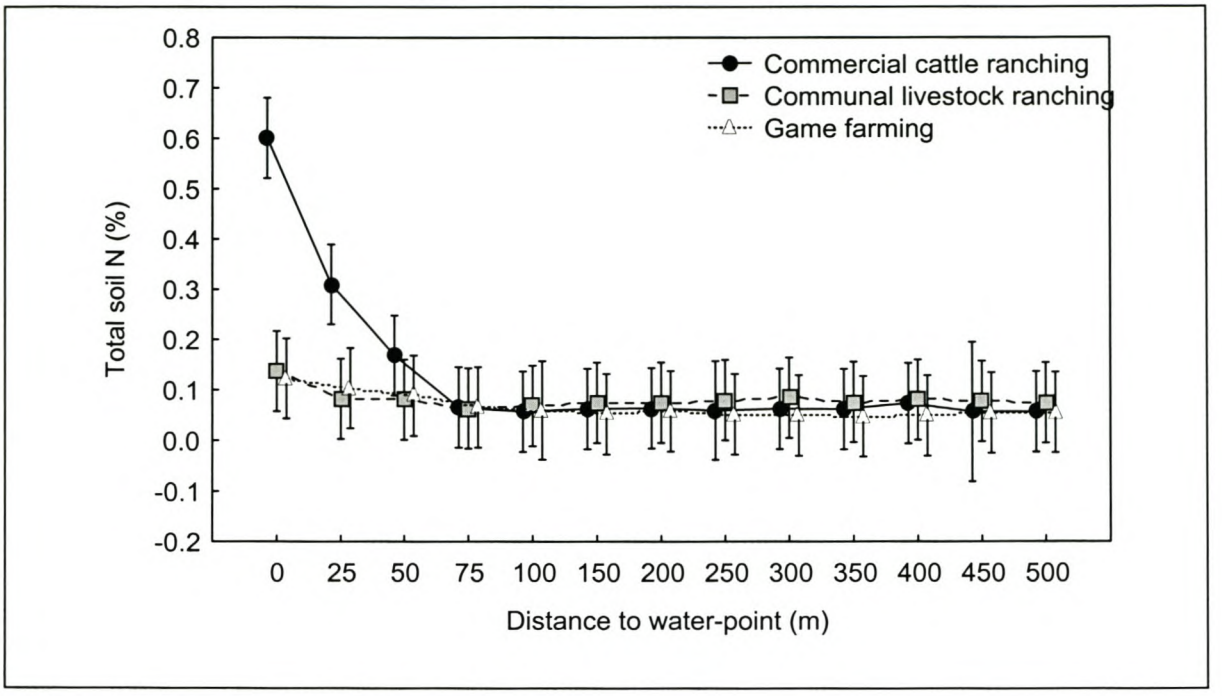

Figure 4: *Total soil N (%) change along transects.*

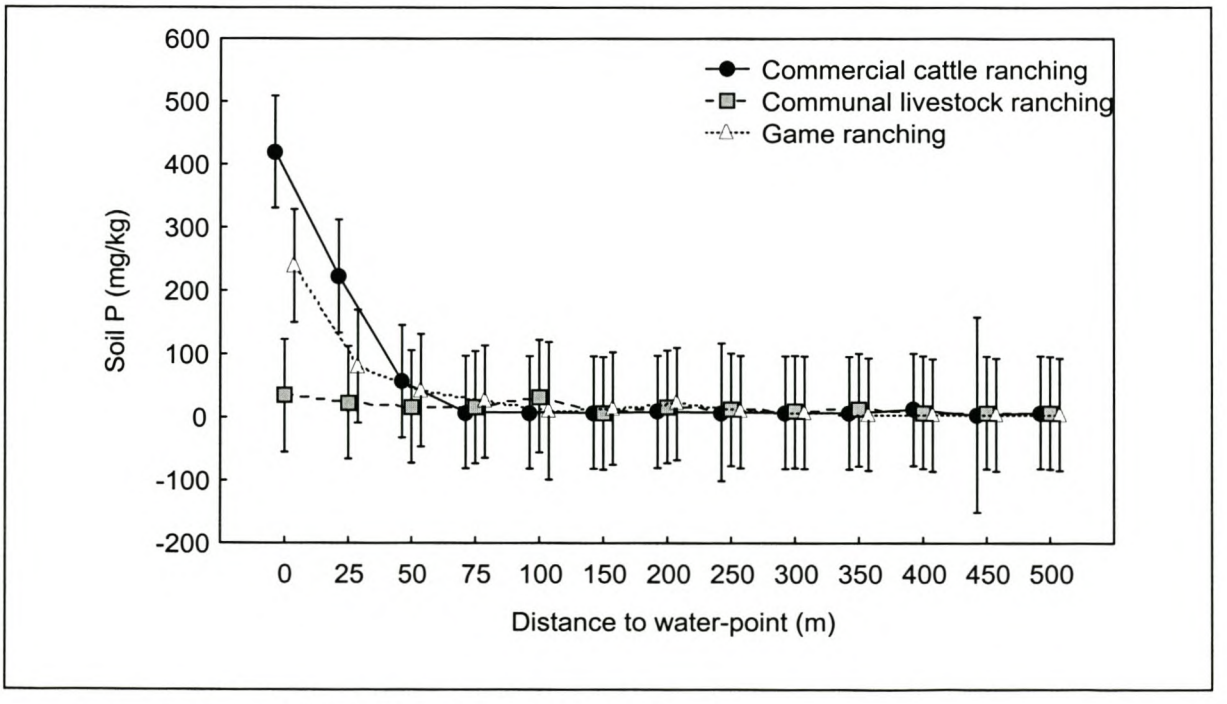

Figure 5: *Available soil P (mg/kg) change along transects.*

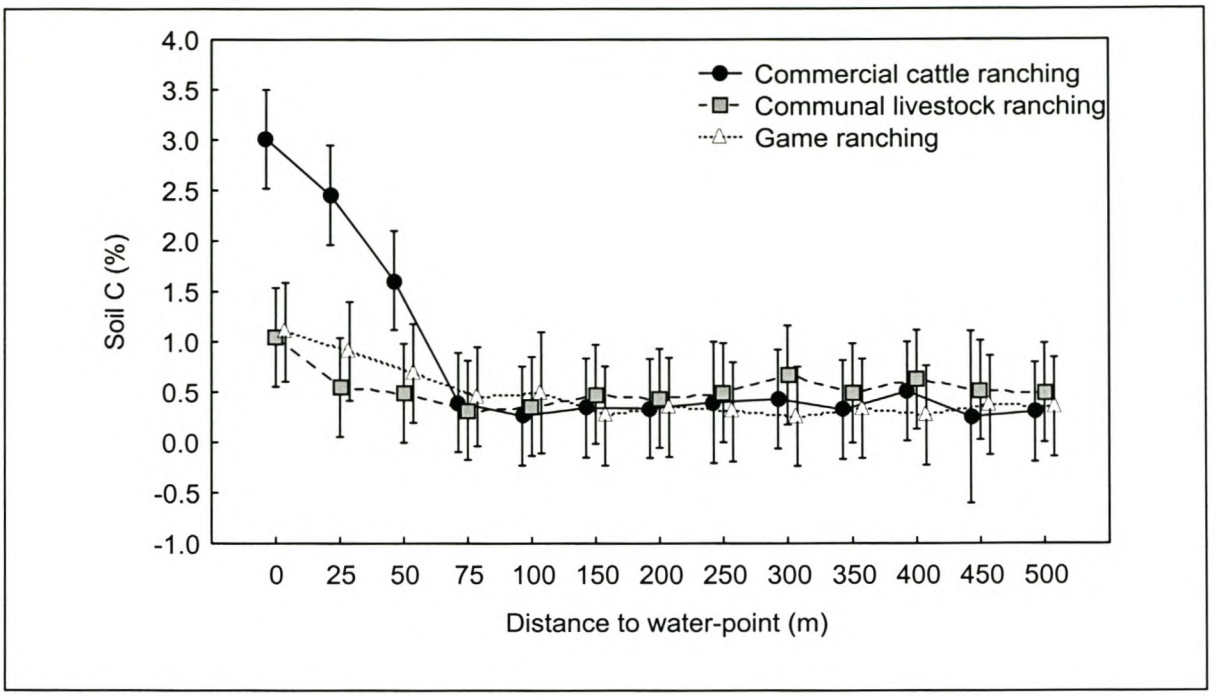

Figure 6: *Soil organic C (%) change along transects.*

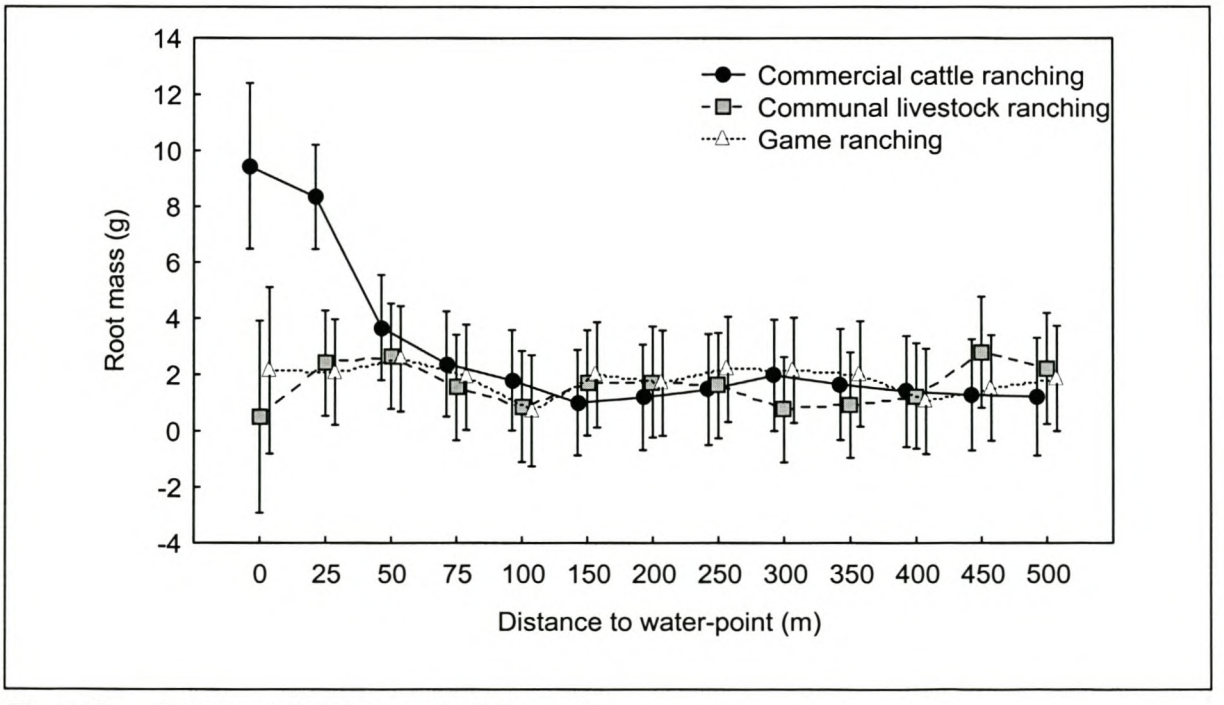

Figure 7: *Root mass (g) change along transects.*

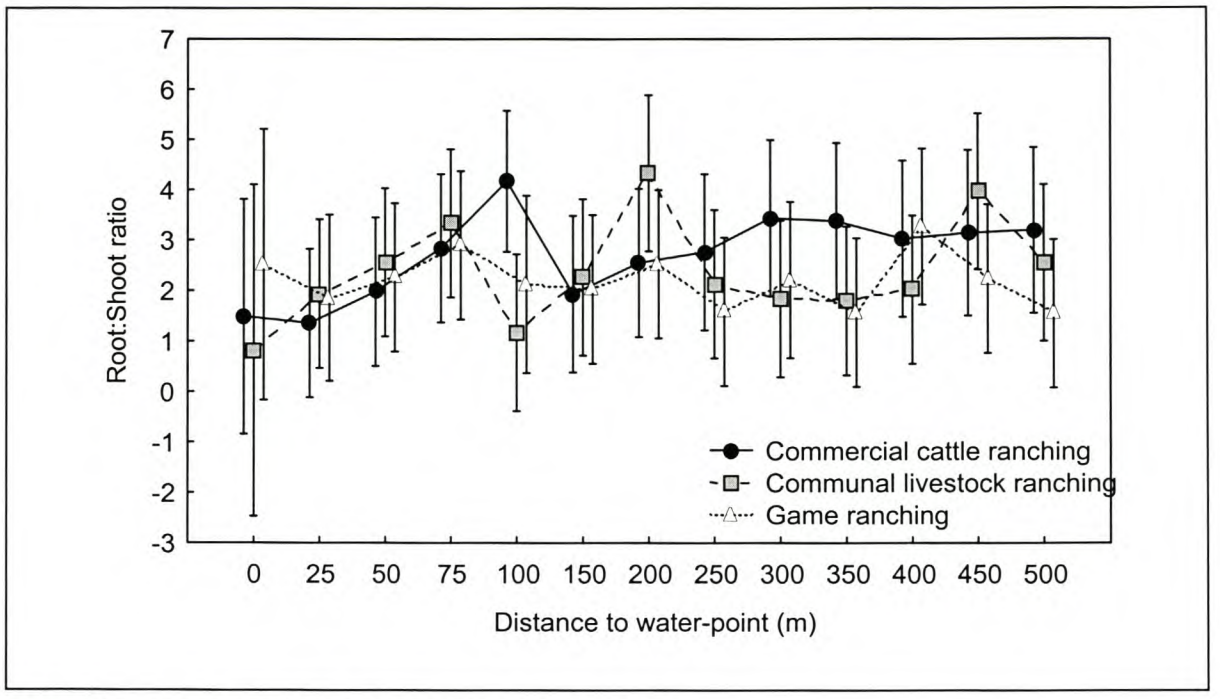

Figure 8: *Root:shoot ratio of bioassays along transects.*

Soil organic carbon

There was a significant effect of management system on soil organic C (ANCOVA: F = 3.782, p = 0.026, df error = 109), as well as an effect of distance on soil organic C (ANCOVA distance: F = 19.136, p < 0.001; Figure 6). Soil organic C was high within 75 m from water-points in the commercial cattle ranches after which soil organic C decreased and stabilised. No effects were observed along the transects of the other two management systems.

Bioassay

Management system did not have a significant effect on shoot dry mass (above-ground mass) (ANCOVA: F = 2.960, p = 0.053, df error = 356), whereas there was an effect of distance on shoot dry weight (ANCOVA distance: F = 22.857, p < 0.001). Shoot dry mass followed similar trends to those for soil nutrients. For commercial cattle ranching, we found high values within 75 m of the water-points, after which mass stabilised at lower values. Under the other two management types, shoot mass remained stable along the entire transect.

Management system did not have an effect on root dry weight (below-ground mass) (ANCOVA: F = 2.522, p = 0.082, df error = 356). There was a significant effect of distance from the water-point (ANCOVA distance: F = 9.063, p = 0.002, df error = 356 – Figure 7). Dry root mass stabilised at 75 m.

Management system did not significantly affect root:shoot ratio (ANCOVA: F 1.876, p = 0.155, df error = 344). There was no effect of distance from the water-point on root:shoot ratio (ANCOVA distance: F = 1.370, p = 0.242 – Figure 8). Figure 8 shows the root:shoot ratio values along transects of the three management types. No pattern in the values of the different management types could be distinguished.

Discussion

Soil quality gradients

Change in soil properties and nutrients generally occurred within 100 m of the water-point. Similar ranges have been recorded by other studies: For example, Tolsma *et al.* (1987) found changes in soil nutrients to occur up to a distance of 100 m from the water-point, while Turner (1998) and Dougill *et al.* (1999) found changes within a distance of 200 m from water-points. Centripetal movement of nutrients towards these zones around water-points has been attributed by Tolsma *et al.* (1987) and Turner (1998) as the reason for the increase in soil nutrients near water-points. All of the soil properties and nutrients can, in some way or other, be influenced by herbivore activity. Herbivore grazing, trampling, defecation, and urination can affect soil pH (Killham, 1994), resistance (Hao & Chang, 2003), N (Whitehead, 2000), P (Scholes & Walker, 1993), and organic C (Snyman, 1999). These properties therefore gave us a good indication that herbivore activity did affect soil quality. Similar trends were found in the vegetation data (see Smet & Ward, 2003).

Commercial cattle ranching

In this study, increased nutrient levels around water-points occurred under commercial cattle ranching in particular. There is a possibility that the increased nutrients close to the water-points are partly derived from the outer regions of the grazing area, as suggested by Tolsma *et al.* (1987) and Turner (1998), through centripetal movement of nutrients. However, a more likely source of nutrients around the water-points is the supplementary feed that is given to cattle at water-points in the commercial cattle ranches. Nitrogen levels around water-points on the commercial cattle ranches were many times higher than those in the communal livestock and game ranches. These particularly high levels are probably partly caused by the urea supplement given to cattle during the dry period as N source (Meissner, 1999). Similarly, low soil resistance might also be caused by the supplementary feeds. However, high salt content in the groundwater might also be a cause of the low resistance.

High organic carbon near water-points is likely to be caused by centripetal movement. Cattle get most of their roughage from vegetation in the grazing areas and regular visit the water-point. Here they deposit organic matter through their faeces. High organic C percentage increases soil fertility (Stewart *et al.*, 1987; Berg *et al.*, 1997; Snyman, 1999) and affects soil pH (Killham, 1994). Soil organic C can buffer acidity and retain a neutral soil pH level (Bloom, 2000).

The soil bioassay recorded a good growth of *Eragrostis tef* within 100 m of the water-point. This is most likely to be caused by the good soil fertility, e.g. high N, P, organic C, and a high pH. Soils near water-points showed low soil resistance indicating elevated salt concentrations. If these salt effects had negatively affected vegetation growth near water-points we were not able to record this. The absence of negative effects of low soil resistance on grass growth is probably caused by the regular watering of the plants in the bioassay, which may have washed out salts.

Communal livestock ranching

Soils around the water-points in the communal livestock ranches did not have low soil resistance, which indicates low soil salinity. This is different from the situation on both commercial cattle and game ranches, which did show low soil resistance around the water-points. This difference is most probably caused by a number of factors: (1) livestock are not supplemented with salts at the water points, (2) water-points are rain-fed, whereas the water-points in the other management systems provide borehole water which is slightly brackish, and (3) because water-points are rain-fed, communal livestock can only use the water-points for a couple of months per year.

When the water-points in the communal ranches run low on water, or completely dry-up, due to drought, livestock rely on water from the bordering Vaal River. Therefore, it is likely that herbivore activity around these water-points is not much higher than in the surrounding areas, except for certain periods when water-points contain water. Thus, distance to water-point has little effect on soil parameters and nutrients. The lower intensity of livestock around the water-points is the most likely reason that soil quality is not affected as much as in the other management systems. On the communal ranches, rangeland degradation might be centred around other key resource areas, such as *Acacia erioloba* woodlands on deep sands or riparian vegetation near the Vaal River. We note in this regard that an early resident of Pniel Estates, Solomon Tshekisho Plaatje, recorded that the *A. erioloba* woodlands were heavily utilized by communally-managed livestock more than 100 years ago, indicating their relative resilience to heavy grazing pressure (Willan, 1996). A more recent study

(Britz, 2003) has shown that communally-managed *A. erioloba* woodlands still produce a high biomass of palatable grasses in spite of prolonged heavy grazing but do have higher abundances of poisonous and unpalatable species than similar woodlands under game ranching.

Game ranching

In contrast to commercial cattle ranching, game ranching had only limited effects on soil properties and nutrients. Game also receive some supplementary nutrients, although in much lower quantities. The water-points that were used were mostly visited by water-independent game that do not stay near the water-point for long periods (Grossman *et al.*, 1999). It is also important to note that the water-points were commercial cattle water-points up to 1989.

Soil P concentration around water-points in the game ranches was higher than under communal livestock ranching and lower than under commercial cattle ranching. The explanation for the high soil P concentrations could lie in the fact that these water-points were used for commercial cattle ranching in the past. Soil P does not easily leach in semi-arid environments and P volatilisation does not occur (Whitehead, 2000). The high soil P around water-points may, therefore, be caused by the previous management system. Total soil N did not show any effect near the water-point. Soil N is very prone to volatilisation and leaching (Scholes & Walker, 1993). Because of this, it is unlikely that high levels of total soil N, accumulated in the soil by commercial cattle ranching in the past, would remain in the soil as long as soil P does. Soil pH around the water-point is still relatively high compared to the communal livestock ranches, which might have to do with a higher percentage of calcrete in the vicinity of the game water-points, and not with increased nutrient levels as might be the case for the commercial cattle ranches. Due to the low availability of nutrients, bioassay results did not differ over the length of the transect, indicating no effect of herbivores.

We may conclude that when commercial cattle ranching is not practiced in the area, soil nutrient levels and soil physical properties will be restored to normal levels. Total soil N and organic C recover in a relatively short time period (< 10 years) and soil P probably over a longer period (> 10 years) due to its low erodibility (Whitehead, 2000).

Rangeland degradation

Our results show that near the water-points there is a clear effect of herbivore activity on soil properties and nutrients, and that management systems seem to have different effects on soil

parameters. Effects among management systems and along the transects can also be seen in vegetation parameters (Smet & Ward, 2003). We therefore conclude that rangeland degradation under the present conditions in the study area is influenced by herbivore activity. We concur with Illius & O'Connor (1999) that the availability of key resources in the area allows for density-dependent coupling between the key resources and the herbivore population.

Due to the availability of key resources in the study area, rangeland degradation is not solely or predominantly caused by non-equilibrium dynamics, as proposed by Ellis & Swift (1988). Rather, equilibrium dynamics seem to have an important role to play in rangeland degradation, affecting soils as observed from our results and vegetation as described in Smet & Ward (2003).

Practical implications

The soil quality gradients around water-points showed that livestock and game had clear effects on soil parameters. Acknowledging that livestock and game affect rangelands has consequences for the way managers manage their rangeland. Unlike Ellis & Swift's (1988) proposal, stocking rate does affect certain parts of the rangeland. Illius & O'Connor (1999) argue that key resources are areas that are primarily prone to degradation due to high stocking rates. It is therefore important for managers to monitor those areas that can be classified as key resource areas. In this study, *Acacia erioloba* woodlands and riparian vegetation along the Vaal River may function as key-resource areas. Areas that are not likely to be key-resource areas are probably less prone to degradation due to lower use of these areas in all but the driest periods and the recharging effect of good rains (Ellis & Swift, 1988; Ward *et al.*, 1998).

Future studies need to determine the events driving ecosystem change in rangelands. This is important as there is no simple criterion that determines whether a system is predominantly driven by biotic or abiotic events (Illius & O'Connor, 1999). Soil quality gradients around key resources are able to provide quick and useful clues to answer part of this question. Measurement of additional soil parameters, such as soil hydrological characteristics, might give an even better understanding of the effect of herbivore activity (such as trampling) on soil condition (Dougill *et al.*, 1999).

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Conclusions and recommendations

Rangeland degradation

Background

Rangeland degradation is a major concern to agricultural and tourism-based industries that rely on the productivity of land resources for production. In addition to directly affecting agricultural production (Abel & Blaikie, 1989), rangeland degradation may also affect biodiversity, primary production, and soil nutrients (Hoffman, 2000). In order to prevent rangeland degradation, a thorough understanding is needed of the system driving rangeland ecosystem change, better known as rangeland vegetation dynamics. Two main theories aim to explain arid and semi-arid ecosystem change:

1. ***Equilibrium theories*** (Clements, 1916). In equilibrium ecosystems, vegetation dynamics are driven by biotic events, such as grazing, in a density-dependent manner. Abiotic events, such as precipitation, control the rate of vegetation change only.
2. ***Non-equilibrium theories*** (Ellis & Swift, 1988). Due to the low and stochastic nature of rainfall in arid to semi-arid environments, water is a limiting factor (Noy-Meir, 1973). In these ecosystems, vegetation change is considered to be driven by rainfall events and not by biotic events.

Traditional rangeland management is based on equilibrium models of vegetation change. Management tools, such as the rangeland condition concept (Dyksterhuis, 1949), and the carrying capacity concept (Stoddart, 1960), influence herbivore populations in order to improve rangeland vegetation. Here, heavy grazing is regarded as the main cause of rangeland degradation (Clements, 1916; Dyksterhuis, 1949; Ellison, 1960; Hardin, 1968). This paradigm remained practically unquestioned until Ellis & Swift (1988) wrote their influential paper on the stability of African arid and semi-arid rangelands (Joyce, 1993). Ellis & Swift (1988), and later Behnke & Scoones (1992), argued that the high stocking rates in communal livestock ranches, which were so often blamed for the bad state of communal lands (e.g. 'the tragedy of the commons' Hardin, 1968), did not lead to degradation of arid and semi-arid rangelands. Their main argument was that in arid and semi-arid environments, the effects of highly variable abiotic events such as precipitation had a much greater influence on vegetation than did grazing. Reducing stocking rates, therefore, did not affect rangeland condition as it was not the cause of degradation. Illius & O'Connor (1999) agreed with Ellis & Swift (1988) that, in certain environments, climatic variability may have a large influence on vegetation dynamics.

However, they argued that certain vegetation patches in these rangelands did show density-dependent processes between vegetation and herbivore populations. Illius & O'Connor (1999) hypothesised that the sizes of herbivore populations were dependent on the availability of certain key-resources. Key resources provide an important continuous source of forage for herbivore populations in highly variable environments (Illius & O'Connor, 1999). Herbivores converge around these resources, which consequently leads to density-dependent processes between herbivore populations and the key-resource vegetation. Degradation of these resources due to heavy grazing can lead to a reduced carrying capacity of the rangeland during dry periods (Illius & O'Connor, 1999).

In short, we can identify three different views on the cause of rangeland degradation:

1. ***Classic, equilibrium view*** - Grazing and other biotic factors are important modifiers of rangeland ecosystems, and vegetation and herbivore populations are coupled in a density-dependent manner. When processes are density-dependent, rangeland managers should manipulate herbivores to ensure minimum degradation and maximum production.
2. ***Non-equilibrium view*** - Low mean and high variability of precipitation in arid and semi-arid environments cause vegetation and herbivore populations to become uncoupled from each other in a density-independent manner.
3. ***Key-resources hypothesis*** - Herbivore populations in arid and semi-arid rangelands are likely to be coupled to key resources in a density-dependent manner. Therefore, large herbivore populations may cause degradation of the key resources. However, vegetation outside the key resources is likely to be driven by rainfall events rather than biotic factors.

The present study compared the effects of three different management systems on vegetation and soil parameters. The three management systems that were compared have significantly different characteristics. They differ in stocking rate, herbivore diversity, grazing regime, and management decision structure. Commercial cattle ranching is characterised by a rotational system in which cattle herds move from one camp to another. The management system applies relatively low stocking rates (16 ha/LSU) and ensures a continuous availability of nutrients through supplemental feeding. Communal livestock ranching does not apply a rotational system, and livestock including cattle, horses, donkeys, goats, and sheep graze and browse the rangeland throughout the year. Stocking rates are quite high (9 ha/LSU) and generally no supplemental nutrients and salts are fed to the livestock. Game ranching consists of more than 17 different game species that continuously graze the

entire rangeland available to them. Stocking rates are high (10 ha/LSU) compared to commercial cattle ranching. During certain periods in the year, game are supplemented with nutrients and salts.

By using 500 m grazing gradients radiating from water-points in the three different management systems, we aimed to determine whether the management systems affect vegetation and soil parameters. Patterns in rangeland degradation should show us whether changes in the rangeland are likely to be driven by climatic (non-equilibrium dynamics) or by biotic events.

Vegetation

Vegetation analysis showed us that certain vegetation parameters were clearly affected by grazing pressure (i.e. distance from water-point). Bare soil frequency and vegetation height were both correlated with grazing pressure (chapter 2). Bare soil frequency was higher near to water-points than it was further away from water-points. For vegetation height we found the reverse, vegetation height was low near water-points and increased with distance from the water-point. We can consider vegetation height as an immediate (rather than a long-term) effect of herbivore activity. This indicator is therefore not a reliable indicator for long-term rangeland degradation. However, bare-soil is a good and reliable indicator of rangeland health (De Soyza *et al.*, 2000).

Plant species composition is an important indicator of long term effects of herbivory. The abundance of perennials and annual grasses is of particular importance (Whitford *et al.*, 1998). We specifically examined the abundance of three grass species:

1. ***Aristida congesta***. This annual or weakly perennial grass species has a low grazing value and is generally found in degraded areas (Van Oudtshoorn, 1999). Furthermore it is considered a increaser II species, i.e. its abundance increases under heavy grazing (Van Oudtshoorn, 1999)
2. ***Eragrostis lehmanniana***. This perennial grass species has average grazing value and an increaser II species (Van Oudtshoorn, 1999). It is well represented in the study area's climax vegetation (Acocks, 1988; Low & Rebelo, 1998).
3. ***Schmidtia pappophoroides***. This grass has a high grazing value is a perennial grass species and considered a decreaser species (Van Oudtshoorn, 1999). This species is also very common to the natural vegetation of the study area (Acocks, 1988; Low & Rebelo, 1998).

We found that only the abundance of *Schmidtia pappophoroides* was positively correlated with distance from water-points. We did not find any effect of grazing intensity on the perennial grass *Eragrostis lehmanniana* and the annual grass *Aristida congesta* between 100 m and 500 m from the water-point. Most of the change in the above-mentioned vegetation occurred within 100 m from the water-point. These results show that within 100 m, herbivore pressure affects vegetation composition. The 100 m zone is comparable with similar zones other studies found in comparable studies (e.g. Tolsma *et al.*, 1987; Andrew, 1988; Jeltsch *et al.*, 1997; Moleele & Perkins, 1998). This zone has been called the *sacrifice zone* (Andrew, 1988), which is often followed by a zone with increased densities of shrubs (Moleele & Perkins, 1998). Our results also indicated the existence of a second zone. When we examine shrub density, we find that there is a positive correlation with distance from water-point. Within 100 m from the water-point, shrub density increased after which it stabilised, which indicates that the denser shrub zone starts beyond the sacrifice zone (James *et al.*, 1999). Tree density was correlated with distance from water-point too. However, here we found a negative correlation. Again, the effect only occurred within 100 m from the water-point, after which no further effect was found.

Vegetation composition was related to management type (chapter 2). Furthermore, the abundance of the three key-grass species showed significant differences among management systems. *Aristida congesta* was more abundant in the communal livestock ranches compared to the other two management systems. Additionally, the two perennial grass species, *Eragrostis lehmanniana* and *Schmidtia pappophoroides*, were least common in the communal livestock ranches. Both commercial cattle ranches and game ranches had significantly less of the annual grass species *Aristida congesta* (which has low grazing value) and more of the perennial grass species *Eragrostis lehmanniana* (high grazing value) along their transects compared to communal livestock ranching. The highest abundance of *Schmidtia pappophoroides* (high grazing value) was found in the game ranches. These results are, therefore, important as they suggest that the heavier stocking rate and selective grazing might have caused a loss of palatable grass species in the communal livestock management systems. We found few signs of degradation in the commercial cattle ranches. This might be due to the relative low stocking rate applied in the ranches (16 ha/LSU). Interestingly, despite the high stocking rate (10 ha/LSU) on the game ranches, this management type showed least degradation along the transects. An important factor that might have contributed to this difference is the high diversity of game (more

than 17 different species) found in the game ranches. Due to the high diversity of game species, the risk of selective grazing is reduced to a minimum (Ritchie & Olff, 1999). Furthermore, some of the game species are so called water-independent species (Grossman *et al.*, 1999). This means that they are able to go without water for long periods and do not visit water-points as often as other species do, which reduces the amount of disturbance around water-points.

Soil

Soils were subjected to two different analysis. All soil samples were analysed using a bioassay in which the grass species *Eragrostis tef* was planted. A sub-sample of the soils was analysed for different soil nutrients (N, P, organic C) and soil characteristics (pH, resistance) (chapter 3). Similar to the vegetation data, soil analysis showed a clear effect of distance to water-points on soil nutrients (chapter 3). Here too, change in the parameters occurred only over the first 75 to 100 m from the water-points, after which soil parameters stabilised. This suggests that soil parameters are influenced by herbivores that gather around the water-point to drink. This in itself is not a surprising result as many have described similar results (Tolsma *et al.*, 1987; Moleele & Perkins, 1998; Turner, 1998). These studies suggest that centripetal movements occur as herbivores graze throughout the rangeland but frequently returning to the water-point where they rest and subsequently defecate and urinate, which increases soil nutrients (Tolsma *et al.*, 1987; Turner, 1998). We concur with Tolsma *et al.* (1987) and Turner (1998) that the difference in soil nutrients and characteristics are most probably partly caused by centripetal movements of nutrients. However, it was interesting to see that soil parameters in the commercial cattle ranches changed dramatically within 100 m from the water-point, whereas soil parameters in the communal livestock ranches were barely affected by distance from the water-point (especially in the case of soil N and P). We believe this is possibly caused by two important differences between the management systems:

1. *Salt and other nutrient supplements* given to animals at water-points. This is a common practice in the game ranches near water-points, but even more common in the commercial cattle ranches, which explains high concentrations of soil nutrients within 100 m from the water-point.
2. Communal livestock water-points that were included in the study are seasonal water-points. This means that activity is not centred around the water-point throughout the year but rather during a limited period when the water-point contains water. For commercial

cattle, the only source of water is the water-point. Therefore, activity is much higher around the water-points.

The results of the bioassay closely followed the results of the soil nutrient and characteristics analysis. This is to be expected as nutrients and characteristics determined plant growth in the bioassay. Water was not a limiting factor in the assay because an automatic sprinkler system ensured a steady water supply. The bioassay yielded no further novel results other than those found in the soil nutrient and characteristics analysis.

Vegetation dynamics

As has been discussed in the previous sections, we found that grazing pressure had less effect on vegetation and soil parameters than management system did. Management system clearly affected vegetation composition, key-grass species abundance, shrub density, tree density, and soil parameters. However, most of the effect was limited within 100 m of the water-point, beyond 100 m soil parameters were barely affected by management system. Like Wiens (1984), we believe that ecosystems are located along a continuum with ecosystems driven by non-equilibrium dynamics on one extreme and those driven by equilibrium dynamics on the other extreme of the continuum. Due to the limitations of the study, we are unable to determine to what extent climatic variability affects vegetation dynamics. However, our data can relate degradation to herbivore activity. Consequently, we can determine to what extent density-dependent processes affects vegetation dynamics.

Illius & O'Connor (1999) suggest that key-resources are likely to be affected by density-dependent processes. We concur with this idea. We believe that water-points function as key-resource areas where density-dependent processes are likely to take place. However, for commercial cattle ranching, water-points are more important areas than for communal livestock or game ranching. Our results suggest that density-dependent processes therefore play a much more prominent role within 100 m from the water-points in the commercial cattle ranches compared to the same zone around water-points in the communal livestock and game ranches. If we look at the overall effect of management systems, we find that communal livestock ranching has less effect on vegetation and soil parameters directly around the water-point compared to commercial cattle ranching. However, beyond 100 m from the water-point, vegetation composition **does not recover** to the degree that it does in the commercial cattle ranches. Therefore, we believe that communal livestock ranching has a greater overall negative effect on vegetation composition. We, however, need to point-out that only the

vegetation data indicated degradation in the communal livestock ranches. Beyond 100 m from the water-points, soil nutrients did not significantly differ among the management systems. This suggests that, while vegetation composition may be affected by management type, this effect may not be irreversible because soil nutrients have not been altered.

Density-dependent processes clearly affect vegetation and soil parameters within 100 m from the water-points. Beyond 100 m from the water-points, we see a clear stabilisation of the measured parameters. This might suggest that density-dependent processes do not affect vegetation dynamics much. However, we cannot conclude that vegetation dynamics are mainly driven by climatic variation. We found clear evidence that management systems affected vegetation composition (chapter 2). This indicates that the different characteristics of management systems do affect vegetation dynamics. However, the degree to which density-dependent processes affects vegetation dynamics differs among them.

Management systems

The three management systems have very different characteristics. In addition to the differences in stocking rate, management systems also differed in management objectives, herbivore diversity, supplemental feeding, decision-making processes, available rangeland, available funds, etc. All these, and other differences, may have led to the different vegetation and soil characteristics we recorded. In the following discussion, the differences, and the effects of the differences between management system are briefly discussed.

Commercial cattle ranching

The commercial cattle ranches mainly produce weaners for the beef industry. Their objectives are therefore to produce as many healthy weaners as possible. This requires cows that are in good condition. Therefore, stocking rates are relatively low (16 ha/LSU) to ensure that there is a continuous supply of forage in the camps. To allow a resting period and good re-growth of the vegetation, herds move from one camp to another in a rotational system. In periods of drought, when forage loses some of its nutritional value, salts and other nutrients are supplied to keep the cattle in optimal condition. Water-points in the commercial cattle ranches are the central points where water and nutrient supplements are supplied to the cattle.

The centralised manner in which water and nutrients are supplied is the main reason for the rangeland degradation within 100 m of the water-points. This zone is called the 'sacrifice zone' (Andrew, 1988) because it is bound to become degraded due to trampling, urination, defecation, and supplement spillage. Beyond 100 m, no relation between distance to water-point and soil parameters was found. This indicates that the severe degradation is limited to a very small area. The direct effects of the sacrifice zone on vegetation were clear from bare soil and vegetation height. These two showed a recovery to normal levels within 100 m from the water-points.

Commercial cattle ranching is characterised by the fact that herds consist of one species. Selective grazing is an often-mentioned risk for areas grazed by single-species herds (Ritchie & Olff, 1999). We have not found any evidence that selective grazing has influenced the vegetation along the various transects. This may be due to the relatively low stocking rates.

Communal livestock ranching

Communal livestock ranching is characterised by having a multi-species herd of which many different people own a few heads of livestock. People belonging to a village have free access to the ranch and are allowed to have their cattle graze its rangeland. This management system has often been criticised, especially by Hardin's (1968) 'tragedy of the commons' in which he reasons that communal livestock ranches are doomed to be overstocked, which will eventually lead to rangeland degradation. However, degradation of communal livestock ranching has become a contentious issue since the publication of Ellis & Swift's (1988) paper discussing the stability of arid and semi-arid rangelands in Africa (see above). Water-points in all management types were chosen to have similar soils, and sites where located within 15 km from each other. This gave us the opportunity to compare the different water-points and management types under similar conditions, hence eliminating the uncertainties brought about by Ellis & Swift's (1988) paper.

Our vegetation analysis indicated that vegetation parameters around water-points in the communal livestock ranches showed more degradation than the other two management systems (chapters 2 & 3). However, this situation may not be a long-term or irreversible situation. Soil parameters did not differ between 100 m and 500 m among the three management systems. Only between 0 m and 100 m, communal livestock ranches soil parameters showed a significant difference from the other two management systems. Here, soil nutrients were less affected by distance from water-points compared to the other management systems. This might have been caused by the fact that most

water-points in the communal livestock ranches are seasonal water-points. These water-points are less likely to be visited often when water is depleted. The Vaal River provides water continuously and is therefore the most important source of water. A likely explanation for the difference in soil nutrients within 100 m of the water-point is the fact that communal livestock do not receive supplementary salts and nutrients at water-points.

The signs of degradation in the vegetation parameters is most likely caused by heavy grazing in the communal livestock ranches. Therefore, we believe that these results most likely indicate the effect of the continuous heavy grazing pressure and selective grazing. However, our results do not necessarily indicate a loss of production in the communal livestock ranches. Communal livestock owners have different objectives for keeping livestock (Behnke & Scoones, 1992; Tapson, 1993). Livestock are used for a myriad of purposes (e.g. traction, dairy products, meat, social exchange) that do not necessarily demand that animals are continuously in good condition.

Game ranching

The game ranches are characterised by the high diversity of game species (over 17 different species). The rangeland is not divided into camps, thereby allowing animals to graze throughout the rangeland. Some supplemental salts and other nutrients are given near water-points.

Game ranching had a positive effect on both vegetation and soil parameters. Abundance of *Aristida congesta* was low and *Schmidtia pappophoroides* abundance was high. The management system also had a positive effect on soil parameters. Distance from water-point does not appear to affect soil N in the game ranches. Soil P, on the other hand, was strongly affected by distance from water-point both in the commercial cattle ranches and the game ranches, although soil P levels in the game ranches were lower (chapter 3). We believe that the relationship between soil P and distance to water-point in the game ranches is a remnant from previous (cattle/sheep) management systems that used the same water-points. The reason that we did not find a similar relation between soil N and distance from water-point may be caused by easy volatilisation and leaching of soil N, whereas soil P is not easily precipitated from the system (Whitehead, 2000).

Recommendations

Concluding, the game ranching seems to have the least negative effects on rangeland condition judging from both our measured vegetation and soil parameters. Although the other two management

systems did affect rangeland vegetation, soil nutrients did not show serious signs of irreversible damage. This means that differences found in rangeland vegetation composition may be of a temporal nature. Productivity of commercial cattle ranches is probably not optimal as cattle only graze grasses. Game and communal livestock ranches can both maintain high stocking rates due to management objectives and herbivore diversity. However, the communal livestock ranches are unlikely to maintain animals in good condition throughout the year. Maintaining animals in good condition may not be necessary for communal livestock ranchers, but it is a necessity for commercial cattle ranchers. Higher stocking rates may not be possible in the commercial cattle ranches due to its objectives. Game ranching should be a more viable option to increase animal production.

The main problem identified around water-points in the commercial cattle management system was the severe degradation within 100 m of the water-point (chapter 2 & 3). This is most likely caused by the frequent visits of cattle to the water-point to drink and feed on the supplementary nutrients and salts. Relieving the water-points can be done is several ways. Supplemental feeding can be done in another part of the camp, e.g. beyond 100 m from the water-point. Another option is to increase the number of water-points so that there are at least two water-points per camp. Water-points can then be used in turns giving vegetation and soil directly around the water-points a longer time to recover.

Communal livestock ranching did not disturb any of the measured soil parameters (chapter 3) as much as commercial cattle ranching did. However, communal livestock management did show to affect the vegetation composition along the entire grazing gradient (chapter 2). The communities are, however, not well organised to take action to control and reduce rangeland degradation. Therefore a group of community members should be chosen to monitor the rangeland and propose plans of action. Management actions proposed by such a group are more likely to be accepted than ideas of individuals. The group may propose temporary exclusion areas as a way to reclaim severely degraded patches around water-points. Restoring old water-points may open up areas that are too far away from the Vaal River.

Game ranching was shown to have a positive effect on rangeland vegetation as well as soil. Severely degraded rangeland around water-points is often caused by the management system used before game ranching started. Reducing the pressure on these areas by closing the water-point may result in improved vegetation and soil parameters. Because game ranching does not apply a camp system, water-points are not restricted to specific areas that optimise the use of the water-point (e.g.

where many camps can have access to the water-point). Water-points should therefore be placed in areas where least damage to soil and vegetation can be incurred. This could be near areas where water-points would form naturally like depressions. The use of mobile water-points may reduce the cumulative effect on vegetation and soil of game drinking at a fixed water-point.

Research methodology

Using 500 m grazing gradients radiating from water-points proved an effective means of assessing how different management systems affect semi-arid ecosystems. The use of grazing gradients is especially useful when the nature of vegetation dynamics is not clear, as is often the case in arid and semi-arid ecosystems (Illius & O'Connor, 1999). This was the case in the study area, as vegetation dynamics can either mainly be driven by climatic variability or by density-dependent processes or both. Water-points provide ideal points to study vegetation dynamics in areas that are, by nature, very spatially heterogeneous – assessing the effects of herbivory in such areas is notoriously difficult because differences between patches can occur for a myriad reasons which may or may not be linked to grazing or browsing. Using water-point-based gradients sidesteps the problem of patchiness and targets the known differences in herbivore activity; near water-points herbivore activity is high and reduces with distance from water-point. Thus, biotic events can be clearly correlated with vegetation change. The length of the transects we used in this study were appropriate to our study area, mainly because the smallest camp did not allow us to increase transect length. Data collection methods such as the point-frequency frame and the point-centred quarter method (Mueller-Dombois & Ellenberg, 1974) were found to be a reliable, and rapid way of collecting large amounts of data in a very systematic way.

The bioassay is a good test of overall soil quality where many different factors might influence plant growth (Olsvig-Whittaker & Morris, 1982). Instead of measuring the level of each individual chemical a bioassay can be a quick and cheap method for examining the effects of different soil treatments on plants. In our case, we did not find a large deviation between the measured soil parameters and the bioassay results. Soil chemical analysis is more accurate and is less dependent on seed quality or other variables that might influence the result but is an indirect means of assessing soil quality as it is not known a priori whether any single nutrient is limiting (e.g. showing that N is lower under communal ranching than under commercial ranching does not necessarily mean that it is low enough to effectively reduce plant growth under communal ranching - Pearcy *et al.*, 1989). Bioassays are

cheaper and therefore useful when a large number of samples is to be analysed (because semi-arid and arid environments are inherently patchy, high replication is a necessity in order to separate effects of herbivory from background levels of soil nutrient variability). We believe that the combination of chemical analysis of a sub-sample of the soils and a bioassay of all the soils is ideal, and increased the reliability and accuracy of the results of this study.

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